Southwest Fisheries Science Center
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STATUS OF THE HAWAIIAN MONK SEAL IN 1992

Timothy J. Ragen

Honolulu Laboratory
Southwest Fisheries Science Center
National Marine Fisheries Service, NOAA
2570 Dole Street, Honolulu, Hawaii 96822-2396

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CONTENTS

INTRODUCTION

The Hawaiian monk seal (*Monachus schauinslandi*) is the most endangered pinniped in U.S. waters. With few exceptions, the present distribution of this species is limited to the Northwestern Hawaiian Islands (NWHI; Fig. 1). Primary breeding populations are found at French Frigate Shoals, Laysan and Lisianski Islands, Pearl and Hermes Reef, and Kure Atoll. Smaller, less productive populations occur at Niihau, Nihoa, and Necker Islands, and at Midway Islands.

The Hawaiian monk seal was first described scientifically by Matschie (1905), who associated this species with the Caribbean monk seal[1] (*M. tropicalis*), which is currently thought to be extinct (Kenyon 1977, 1980; LeBoeuf et al. 1986), and the Mediterranean monk seal (*M. monachus*), which is now severely endangered. Historical patterns of abundance and distribution are poorly documented for the Hawaiian species. However, existing records are sufficient to suggest two periods of significant decline: the first resulted from sealing activities in the 1800s and the second occurred between the late 1950s to the 1970s. In spite of extensive management efforts since the late 1970s, the Hawaiian monk seal has not recovered. In the past 3 years the species has declined further, primarily due to decreases in birth rate and juvenile survival of seals at French Frigate Shoals.

History of Exploitation and Management

Written reports of the Hawaiian monk seal begin with the Russian explorer Lisianski (1814), who, in 1805, observed seals on the island that now bears his name. Records from voyages of the *Aiona* in 1824 (Bryan 1915) and the *Gambia* in 1859 (The Polynesian, 13 August 1859) suggest that a period of sealing occurred in the early to mid 1800s. Presumably, seals were killed for oil and pelts, but they were also killed for food by crews of wrecked vessels and by guano and feather hunters (Dill and Bryan 1912, Wetmore 1925, Clapp and Woodward 1972).

The effect of sealing on total abundance and distribution is unknown, but probably was severe. The *Gambia* reportedly returned to Honolulu with 1,500 skins (although the authenticity of this report has been questioned; see Kenyon and Rice 1959). By the late 1800s, sightings of monk seals were rare, suggesting their numbers had been decimated. For example, no seals were seen at Midway over a period of 14 months in 1888-89, and only one seal was seen on Laysan Island during 3 months of observation in the

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[1] Repenning and Ray (1977) suggest that the Hawaiian monk seal may have originated from a Caribbean ancestor more than 15 million years ago. Because it retains a number of primitive characteristics, they referred to this species as a living fossil.

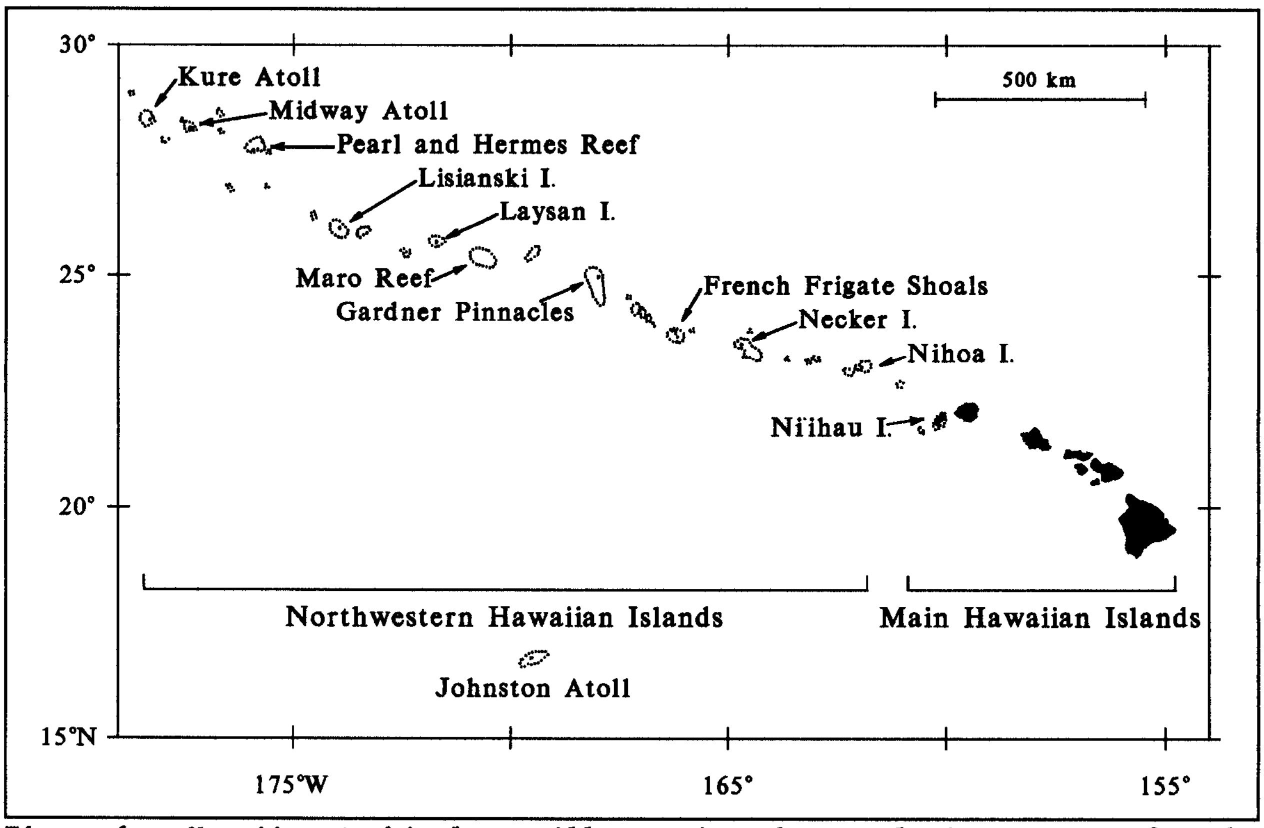

Figure 1. Hawaiian Archipelago, illustrating the population centers for the Hawaiian monk seal. The dotted lines around the Northwestern Hawaiian Islands indicate the 100-fa isobath.

winter of 1912-13 (Bailey 1952).

If the Hawaiian monk seal was decimated by the late 1800s, then during the first half of this century the species must have experienced at least partial recovery. Unfortunately, this recovery was poorly documented (Svihla 1959, Rice 1964, Hiruki and Ragen 1992). The first reliable range-wide surveys were conducted in the late 1950s (Kenyon and Rice 1959, Rice 1960), and additional counts were conducted at Midway Islands in 1956-58 (Rice 1960) and at Kure Atoll in 1963-65 (Wirtz 1968). Surveys were repeated throughout the 1960s and 1970s, but the results are difficult to compare because the survey methods were not standardized with respect to the population(s) counted, season, time of day, count method, differentiation between size and sex classes, and extent (partial or full island or atoll count).

In spite of the lack of standardization, sufficient information was gathered to demonstrate a decline of approximately 50 percent in the total number of Hawaiian monk seals between the late 1950s and the mid-to-late 1970s (Kenyon 1973, Johnson et al. 1982). The decline occurred in the western portion of the species' range, and involved populations at Kure

Atoll, Midway Islands, Pearl and Hermes Reef, and Lisianski and Laysan Islands. Contrary to the trend at other locations, the French Frigate Shoals population grew substantially during this period.

These changes have not been explained for populations at Pearl and Hermes Reef, or at Lisianski and Laysan Islands. However, at Midway and Kure Atolls, and at French Frigate Shoals the trends have been associated with changes in the level of disturbance by U.S. Navy and Coast Guard personnel stationed at those locations (Kenyon 1972, 1980; Johnson et al. 1982; Gerrodette and Gilmartin 1990). At Kure, for example, Kenyon (1972) observed that pregnant females abandoned preferred pupping locations. He attributed their behavior to increased disturbance from human activity. The apparent consequence was a reduction in pup survival, which led to poor recruitment and the eventual reproductive failure of the Kure population.

As a result of the general decline, in 1976 the Hawaiian monk seal was designated as depleted under the Marine Mammal Protection Act of 1972 (41 FR 30120) and endangered under the U.S. Endangered Species Act of 1973 (41 FR 51611). Shortly thereafter, the Marine Mammal Commission, the National Marine Fisheries Service (NMFS), and the Fish and Wildlife Service (FWS) began to investigate the natural history of this seal and its decline.

These efforts were directed toward two objectives. First, in 1978 the Marine Mammal Commission sponsored a workshop to consider future research on the Hawaiian monk seal. The workshop resulted in a 5-year workplan listing important research questions to be addressed (Kenyon 1978). These questions were considered again in the subsequent development of the recovery plan for the Hawaiian monk seal (Gilmartin 1983).

Second, field studies were conducted in 1977-80 at Laysan Island (Johnson and Johnson 1978, 1981a, 1981b, 1984) and Kure Atoll (Johnson et al. 1982). These studies provided extensive information on the life history, behavior, and status of monk seal populations at those sites. Importantly, Johnson and Johnson (1981a, 1984) documented a die-off of seals at Laysan Island in the spring of 1978. They estimated that at least 50 seals died during this episode. The cause could not be conclusively determined, but Gilmartin et al. (1980) found evidence indicating the seals may have succumbed to ciguatoxin or maitotoxin. The Marine Mammal Commission sponsored a workshop in April 1980 to review findings of the investigation into this die-off and to develop a contingency plan for responding to future episodes of mass mortality (Gilmartin 1987).

In 1980, NMFS began its monitoring program at Lisianski Island. In the same year, the Hawaiian Monk Seal Recovery Team met to develop a recovery plan, and NMFS completed a draft environmental impact statement on the designation of critical habitat for the species. The statement proposed that critical habitat extend out to the 10-fathom (fa) isobath adjacent to pupping and haulout islands.

In 1981, NMFS extended its monitoring to Laysan Island and Kure Atoll, and initiated a program to restore the severely depleted population at Kure (Gilmartin et al. 1986). The objective of this "Head Start" program was to enhance the survival of young females and thereby increase subsequent recruitment into the adult female population. Weaned female pups were captured and temporarily held in a shoreline enclosure, where they were protected from sharks and aggressive adult males while they developed independent feeding skills.

In the same year, NMFS reviewed and submitted a biological opinion on the "Combined Fishery Management Plan, Environmental Impact Statement and Regulatory Analysis for the Spiny Lobster Fisheries of the Western Pacific Region." The lobster fishery began in the NWHI during the late 1970s, and created a potential source of direct and indirect fishery interactions with protected species such as the Hawaiian monk seal and the green turtle (*Chelonia mydas*). To minimize interactions, this plan prohibited fishing in waters less than 10 fa deep in the Fishery Conservation Zone (FCZ) of the NWHI, required permits for fishing and reports of catch and effort data, and provided a mechanism for evaluating and responding to interactions involving monk seal mortality. (Other actions and amendments of this plan are listed in Appendix A.)

In 1982, NMFS began annual monitoring of Hawaiian monk seal populations at French Frigate Shoals and Pearl and Hermes Reef. In addition, all seals at Lisianski Island were bleach marked to determine haulout patterns, and a study was initiated to determine the effects of flipper-tagging on recently weaned pups (Henderson and Johanos 1988). In the following 2 years the tagging program was expanded to the other main breeding populations and thereafter, most weaned pups have been tagged. The identification of tagged individuals has been a critical element in the study of monk seal life history traits and population dynamics.

The "Recovery Plan for the Hawaiian Monk Seal, *Monachus schauinslandi*" was completed in 1983 (Gilmartin 1983). The plan emphasized (1) identification and mitigation of factors causing decreased survival and productivity, (2) characterization of habitat, including foraging areas, (3) assessment and monitoring of population trends, (4) documentation and mitigation of negative effects from human activities, (5) implementation of conservation-oriented management actions, and (6) development of educational programs to enhance public conservation efforts.

The following year, 1984, a rehabilitation program was started to further enhance recovery at Kure Atoll (Gerrodette and Gilmartin 1990). Undersized, weaned female pups were taken from French Frigate Shoals to Oahu, where they were held in captivity for 8-10 months to increase weight and develop feeding skills. At the end of the captive period these animals were screened for disease and transported to the shoreline enclosure at Kure Atoll. They were released into the wild at Kure after they had demonstrated that they were able to sustain themselves by feeding on live fish.

Two important management plans were proposed in 1984. A Supplemental Environmental Impact Statement was prepared by NMFS to designate critical habitat for the Hawaiian monk seal. As in the Draft Environmental Impact Statement, the proposed critical habitat included the major hauling and pupping islands and surrounding waters out to the 10-fa isobath. Also, a draft Master Plan for the Management of the Hawaiian Islands National Wildlife Refuge was completed by the FWS; this plan encompassed the majority of islands inhabited by the Hawaiian monk seal.

In 1986, NMFS prepared a biological opinion for the "Combined Draft Fishery Management Plan, Environmental Assessment, and Regulatory Impact Review for the Bottomfish and Seamount Groundfish Fisheries on the Western Pacific Ocean." Among other actions (see Appendix A), this plan prohibited the use of bottom trawl, bottom-set gillnets, explosives, and poisons in the FCZ, and established a permit requirement for fishing for bottomfish in the FCZ of the NWHI (with an allowance for experimental fishery permits). As with the lobster fishery, the bottomfish and seamount fisheries created a potential source of direct and indirect fishery interaction with monk seals. In 1986 the Marine Mammal Commission recommended that NMFS amend the plan to provide for monitoring and verification of interaction rates with Hawaiian monk seals and other protected species. Amendment #4 of this plan established Protected Species Zones extending out 50 nautical miles (nmi) from the NWHI and gave NMFS authority to place observers on vessels planning to fish within these zones; hence, this amendment provided a mechanism to determine the rate of interaction of protected species with fishing gear and operations of the bottomfish and groundfish fisheries.

In the same year, NMFS submitted a biological opinion on the "Fishery Management Plan for the Pelagic Fisheries of the Western Pacific Region" (Appendix A). This plan prohibited foreign longline vessels from fishing within 100 nmi of the NWHI. Foreign longliners wishing to fish in the remaining open areas of the FCZ were required to submit effort plans, obtain permits, carry observers when requested, and report data on catch, effort, and interactions with sea turtles and marine mammals. The plan prohibited all drift gillnetting in the FCZ except for fishing by domestic vessels with an experimental fishery permit. Vessels with permits were required to collect and submit data on catch, effort, and interactions with sea turtles and marine mammals.

Critical habitat was designated in 1986 as all beach areas, lagoon waters, and ocean waters out to a depth of 10 fa around Kure Atoll, Midway Islands (except Sand Island), Pearl and Hermes Reef, Lisianski Island, Laysan Island, Gardner Pinnacles, French Frigate Shoals, Necker Island, and Nihoa Island (Federal Register, 30 April 1986, 51FR16047). However, because of concerns raised by the Marine Mammal Commission and the threat of legal action by the Sierra Club Legal Defense Fund, NMFS reopened the comment period on critical habitat which, in 1988, was extended to include Maro Reef and waters around existing habitat out to the 20-fa isobath (Federal Register, 26 May 1988, 53FR18988).

The problem of male Hawaiian monk seal "mobbing" behavior was discussed in a workshop held in 1987 at the NMFS Honolulu Laboratory (Gilmartin and Alcorn 1987). Mobbing occurs when multiple males attempt to mate simultaneously with a single victim, usually (but not always) an adult female. This behavior was first observed in 1978 (Johnson and Johnson 1981a), but injuries which may have resulted from such behavior were noted by Walker (1964), Kridler (1966), Wirtz (1968), Olsen (1969) and DeLong et al. (1976).

In 1989 the Hawaiian Monk Seal Recovery Team met for the first time since 1984, and recommended priorities for a 3-year work plan of recovery actions for the species (Gilmartin 1990). The plan emphasized the importance of (1) mitigation of the effects of mobbing behavior at Laysan and Lisianski Islands, (2) continued monitoring of the 5 main breeding populations, and (3) facilitating the recovery of the populations at Kure Atoll, Midway Islands, and Pearl and Hermes Reef.

Addressing these priorities has remained a management goal. However, in the past 4 years extensive management effort has been directed toward monitoring and mitigating new threats to the species. The pelagic longline fishery in Hawaiian waters grew from approximately 37 vessels in 1987 to approximately 140 vessels in 1990 (Ito 1992). This rapid growth raised concern that monk seal interactions with longline operations would increase. In May 1990, unconfirmed reports suggested such interactions were occurring. In August 1990, the Western Pacific Regional Fishery Management Council proposed an emergency action under its Bottomfish and Pelagic Fisheries Management Plans, and in November 1990 NMFS published an emergency ruling (55 FR 49285) which required that (1) longline vessels obtain permits from NMFS (permits were already required for vessels fishing for bottomfish), (2) both longline and bottomfish vessels provide daily logs with information on their interactions with monk seals and other protected species, (3) these vessels notify NMFS prior to fishing within 50 nmi of the NWHI to provide the Regional Director the option of placing an observer on board, and (4) longline operators attend an orientation meeting to learn about procedures for protecting endangered and threatened species. By January 1991 seals at French Frigate Shoals were seen with injuries or embedded hooks, providing further evidence of interactions. In April 1991, NMFS published emergency rules establishing a moratorium on pelagic longline fishing within a Protected Species Zone extending 50 nmi around the NWHI and in the corridors between islands. These rules were made permanent in October 1991 (56 FR 51849 and 56 FR 52214, respectively).

In the same period, 1989-92, other significant and alarming changes occurred. In 1990 the birth rate at all major breeding islands was extremely low; only 28 percent of the adult females at Laysan Island gave birth in that year. In 1991, birth rates returned to more normal levels at all locations except French Frigate Shoals, where the number of births declined to the lowest level since extensive monitoring began in the early 1980s. Survival rates, particularly for juveniles, also declined during this period and, as was the case for reproductive rates, the low survival rates persisted at French Frigate Shoals. Furthermore, many surviving juveniles at French Frigate Shoals were in poor condition or emaciated.

Due to the severity and persistence of the changes observed at French Frigate Shoals, management efforts were directed primarily at this population. These efforts focused first on evaluation of the problem (Gilmartin and Ragen 1992) and then on rehabilitation of juvenile females either in poor condition or small for their age. In 1992, 20 seals were transferred from French Frigate Shoals to Midway, where historical data and the general condition of seals indicate the environment should support more seals. Three of those seals were temporarily returned to Oahu for more intensive care; these three improved quickly and, in early January 1993, were taken to Midway and released.

Life History Information

Because Hawaiian monk seal populations are small, the investigation of their life history has relied on the identification and study of individual animals. Temporary identification (between annual molts) has been achieved primarily through the application of commercial bleach to the animal's pelage (Johnson and Johnson 1978, 1981a; Stone 1984). Permanent or longterm identification has been based on natural features, such as scars or pelage marks, and on the application of a variety of tags. Tags were first applied by the FWS in 1966, and by NMFS in 1981 at Kure Atoll (Gilmartin et al. 1986). Since 1984, tags have been applied to most weaned pups in all the main populations. Monitoring of these tagged animals is beginning to provide information on age- and sex-specific life history parameters and patterns.

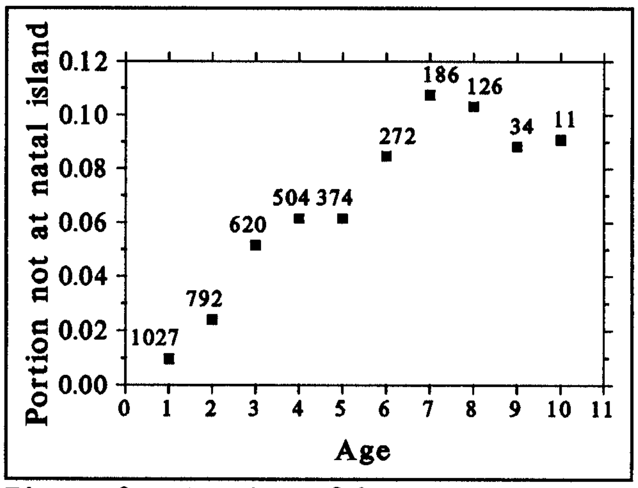

Figure 2. Portion of known-age animals sighted at locations other than their natal island. Numbers are samples sizes.

As with other pinnipeds, the behavior of Hawaiian monk seals is most easily studied when they haul out on land; virtually nothing is known of monk seal behavior at sea. Hawaiian monk seals are usually solitary animals, associating with each other primarily during their protracted reproductive season. In general, they show a high degree of fidelity to their natal island or atoll and do not migrate. Johnson and Kridler (1983) reviewed resighting records for 351 seals tagged between 1966 and 1972, and found that 3 to 15 percent of seals moved between islands. Similarly, of seals tagged as pups by NMFS, the number sighted at locations other than their natal island increases with age to approximately 10 percent by age 6-10 (Fig. 2, NMFS unpubl. data). At a particular site, haulout patterns are known to vary widely among individuals and by size and sex class (Johnson and Johnson 1978, 1981a, 1981b, 1984; Stone 1984, Johanos et al. 1986), season (Johnson and Johnson 1984), and reproductive state (parturient versus nonparturient; Johanos et al. in press). Because some populations occur at single islands (i.e., Laysan and Lisianski Islands) and others occur at multi-island atolls (i.e., French Frigate Shoals, Pearl and Hermes Reef, Midway Islands, and Kure Atoll) haulout patterns are also influenced by the geography of the home island or atoll.

Monk seals haul out on land to rest, interact with other seals (primarily during the reproductive season), give birth and nurse their young, and molt. Molting occurs annually, primarily between April and December (Johnson and Johnson 1978, 1981a, 1981b, 1984); the actual timing for individual animals varies by age and sex class (Kenyon and Rice 1959; Johnson and Johnson 1978, 1981a, 1981b, 1984) and reproductive status (Johnson and Johnson 1978, 1981a, 1981b, 1984, Johanos et al. in press). For an individual seal, visible changes occur, on average, over a period of 9-10 days (Kenyon and Rice 1959, Johnson and Johnson 1984). With the exception of the first postnatal molt, which requires several weeks and begins late in the nursing period, the molt includes the pelage and the epidermis (Kenyon and Rice 1959).

In addition to molting, many activities related to reproduction occur on land, and reproductive patterns and behaviors have been studied extensively. Interactions between adult males and association patterns of adult males with adult females suggest that reproductive access to females is determined by a dominance hierarchy among males (NMFS unpubl. data). The breeding season is protracted and estrus in adult females is relatively asynchronous (Atkinson and Gilmartin 1992, Johanos et al. in press); hence, dominant males are able to control access to, and presumably mate with, multiple females.

Major events in the annual reproductive cycle of adult females at Laysan Island are described in Johanos et al. (in press). The relative timing of events for Laysan females is probably representative of female patterns at other locations. However, the reproductive seasons at different islands may be slightly out of phase. For example, there is some indication that, on average, births at French Frigate Shoals occur later in the spring than at Kure Atoll or Laysan Island (Johnson and Johnson 1980, NMFS unpubl. data). Births have been observed in all months of the year (W.G. Gilmartin, pers. comm.), but most occur between February and August with the peak in April and May. Pregnant females prefer selected sites for parturition (Westlake and Gilmartin 1990); they may haul out several days before pupping. Newborn pups of both sexes are black and weigh approximately 15-17 kg (Kenyon and Rice 1959). On average, pups nurse approximately 5 to 6 weeks (Kenyon and Rice 1959, Johnson and Johnson 1978, 1984; Boness 1990; Johanos et al. in press) and gain 50-80 kg. Mothers fast and remain with their pups throughout the nursing period. During this period, mothers are generally intolerant of other seals, including other mother-pup pairs. Pup switches are more likely to occur when mother-pup pairs are in close proximity. Kenyon and Rice (1959) noted that females may have trouble distinguishing their own pups, but Johnson and Johnson (1978) were the first to document that such confusion often led to the switching of pups. While such switching does not appear to be common in other pinnipeds, Boness (1990) was unable to detect any selective disadvantage with respect to pup survival. Also, a mother will foster another pup if her own dies or is lost (Gerrodette et al. 1992). Weaning occurs when the mother abandons her pup.

Mating is aquatic and rarely observed, but often results in characteristic injuries on the female's dorsum. The timing and incidence of these injuries indicates that the interval between weaning and mating is highly variable, but that on average, parturient females mate 28 days (± 21 days, SD) after weaning (Johanos et al. in press). Nonparturient females mate (on average) approximately 2 weeks earlier in the reproductive season. The study of monk seal mating behavior is particularly important because population recovery at several locations appears to be inhibited by the loss of females from "mobbing," which occurs when multiple adult males attempt to mate simultaneously with the same seal. The mobbed seal is most often an adult or subadult female, but juvenile females and males of all sizes are also mobbed (Hiruki et al. in press [a], Hiruki et al. in press [b]).

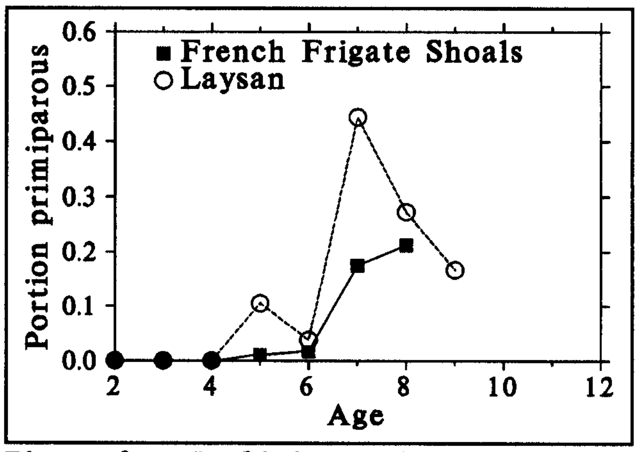

Figure 3. Preliminary data on age of first birth at Laysan Island and French Frigate Shoals.

Reproductive rates for Hawaiian monk seal females vary substantially, both by individual and by year (Johanos et al. in press). Because relatively few known-age animals (i.e., those tagged as pups between 1982-92) have reached maturity, age-specific birth rates are not yet available. Preliminary data suggest that the mean age of first birth is approximately 6 to 7 years, but may vary by island population (Fig. 3). At Laysan Island between 1982 and 1991, the pooled reproductive rate was 0.546 (n = 491) for all adult-sized females and 0.675 for females known to be mature (i.e., known to have given birth in a previous year; Johanos et al. in press). The mean annual rates for these two groups were 0.558 (± 0.126, SD; n = 9 years) and 0.672 (± 0.111, SD; n = 10 years). As Figure 4 illustrates, the annual variation in these rates was large.

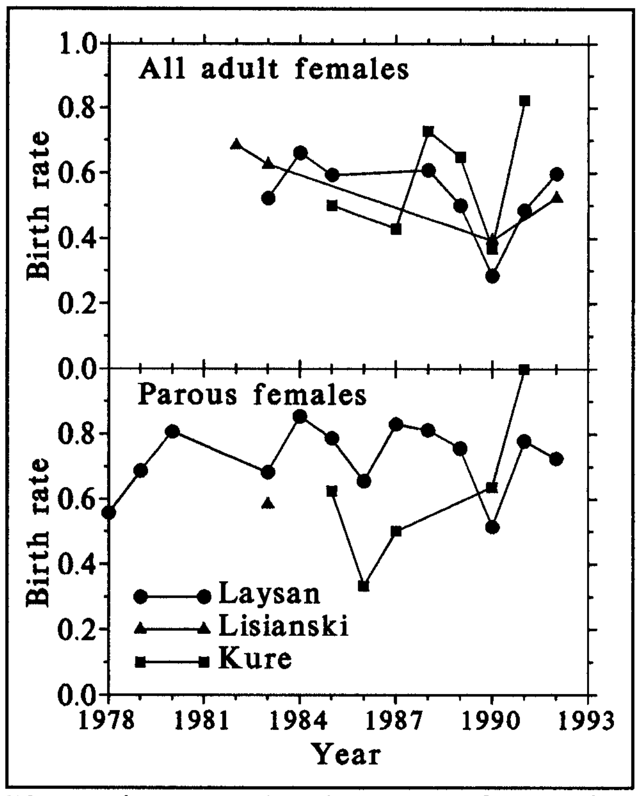

Figure 4. Reproductive rates for adult-sized females (top panel) and females known to have given birth in a previous year (bottom panel); data are from Laysan and Lisianski Islands and Kure Atoll.

Survival rates are similarly variable. While this variability is most apparent in younger age classes, survival of older age classes can also vary dramatically. As noted earlier, Johnson and Johnson (1981a, 1984) estimated that at least 50 seals died at Laysan Island in 1978; the cause(s) was uncertain, but may have been due to poisoning by ciguatoxin and maitotoxin (Gilmartin et al. 1980). Other known sources of mortality for all ages include sharks (particularly the tiger shark, *Galeocerdo cuvier*) (Balazs and Whittow 1979, Alcorn and Kam 1986), injuries inflicted by adult males (Banish and Gilmartin 1992, Hiruki et al. in press [a], Hiruki et al. in press [b]), disease (Banish and Gilmartin in press), entanglement in marine debris (Henderson 1985, 1990), and starvation (primarily affecting younger seals; Banish and Gilmartin 1992). The relative significance of each of these sources of mortality is undetermined. Sufficient data on age-specific survival rates of adults are not yet available, but are accumulating as known-age (tagged) seals mature.

Gilmartin et al. (in press) estimated first-year survival for the pups of the five major breeding populations in the period 1982-87. Their results indicate that survival from weaning through the first year of life is approximately 0.8 to 0.9, increasing thereafter to 0.9 or greater. They also found that survival varied by island, by sex (females may experience better survival than males), and by year.

In addition to these sources of variability, survival (at least during the first year) also appears to vary by size at weaning (Fig. 5, NMFS unpubl. data). The relationship between axillary girth (at weaning) and survival has been used to identify pups that have less chance of survival and are therefore more likely to benefit from rehabilitation and relocation into a favorable environment. Hence, the rehabilitation and transfer of seals from French Frigate Shoals to Kure Atoll (1985-91) or Midway Islands (1992) has focused on female pups considered to be undersized on the basis of their axillary girth at weaning.

As noted earlier, newborn pups weigh approximately 15-17 kg and are about 1 m long (Kenyon and Rice 1959; these authors did not measure axillary girths). Growth is extensive during the nursing period and at French Frigate Shoals (1990-92), for example, mean weight for a newly weaned pup was 62.7 kg (± 15.9 kg, n = 167), mean length was 125.9 cm (± 7.7 cm, n = 266), and mean axillary girth was 102.7 cm (± 10.5 cm, n = 267; M. Craig, pers. comm.). In the months after weaning pups lose much of their weight and girth while making the transition to independent feeding. At the

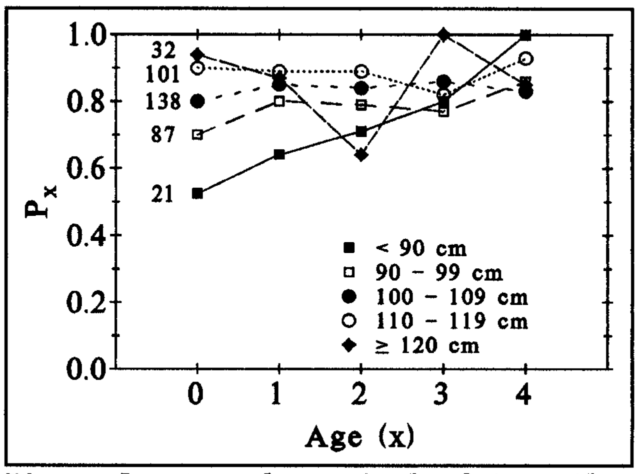

Figure 5. Annual survival of juveniles at French Frigate Shoals as a function of their axillary girth at weaning. Numbers are sample sizes.

end of the first year, their lengths will have increased approximately 10 cm, but their weights and girths will have decreased by approximately 10 kg and 10 cm, respectively (M. Craig, pers. comm.). Thereafter, size probably increases monotonically until adulthood (with the exception of fluctuations in weight and girth related to the annual molt). Few adults have been weighed or measured, but Rice (1964) suggests that on average, adult females weigh approximately 205 kg and are about 2.3 m in length, while the average adult male is smaller at about 170 kg and 2.1 m.

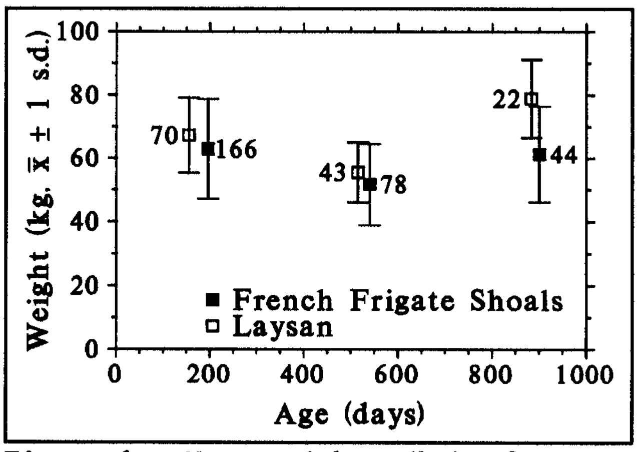

Figure 6. Mean weights (kg) of weaned pups, and 1- and 2-year-olds at French Frigate Shoals and Laysan Island, 1990-92. Age is days after January 1 of the year each seal was born. Numbers are sample sizes.

Because young seals appear to be more tolerant of handling and can be measured more efficiently, NMFS studies of Hawaiian monk seal growth have been limited to animals less than 3 years of age. The primary goal for these studies has been to determine if growth patterns or rates have changed at French Frigate Shoals in a manner consistent with the hypothesis that abundance at this site has approached or reached the environment's carrying capacity (K). Preliminary results support this hypothesis; comparisons of weaned pups and juveniles from French Frigate Shoals with seals of the same age from Laysan Island indicate that growth has been slower at French Frigate Shoals (Fig. 6; M. Craig, pers. comm.). In Figure 6, note that (1) for each age group, mean weights were smaller at French Frigate Shoals, and (2) the largest difference in weight occurred in 2-year-old seals. (A statistical analysis of these data is being completed and will be reported elsewhere.) Note also that the slower growth at French Frigate Shoals is consistent with recent (1990-92) observations that a relatively large portion of seals at that site are in poor condition.

Growth rates must vary as a function of food availability. The assumption that slow growth rates indicate the French Frigate Shoals population is near K implies that increasing monk seal density leads to a reduction in prey abundance or availability to individual seals; i.e., that food is an important limiting factor. This reasonable hypothesis has been difficult to test because the food habits of Hawaiian monk seals are poorly known. The majority of information on their food preferences comes from collected scats and spews (DeLong et al. 1984, NMFS unpubl. data), which indicate that monk seals are opportunistic feeders preying on octopus, squid, and lobster, and a number of reef and benthic fishes. Fishes more commonly found in scats and spews include triggerfish (Balistidae), butterflyfish (Chaetodontidae), conger eels (Congridae), squirrelfish (Holocentridae), rudderfish (Kyphosidae), wrasses (Labridae), goatfish (Mullidae), moray eels (Muraenidae), brotulas and cusk eels (Ophidiidae), damselfish (Pomacentridae), bigeyefish (Priacanthidae), parrotfish (Scaridae), and lizard fish (Synodontidae). Fishes occurring less commonly in scats and spews include hawkfish (Cirritidae), angelfish (Pomacanthidae), flagtailfish (Kuhliidae), scorpionfish (Scorpaenidae), jacks (Carangidae), and mullets (Mugilidae). However, scats and spews are known to overemphasize the importance of nearshore prey with hard tissues that are passed through the digestive tract. Food items eaten farther offshore or with no lasting hard tissues are less likely to be represented. Furthermore, without information on the animal leaving the scat or spew, the recovered materials cannot be used to investigate foraging differences among the various size and sex classes. The poor condition of juvenile seals underscores the importance of evaluating their foraging preferences and distributions.

Studies of monk seal prey distributions have focused on the depth of prey as inferred from seal diving patterns. Three studies of monk seal diving depths are summarized in Table 1. Their results are consistent with shallow nearshore feeding, but also illustrate the ability of monk seals to dive and forage in deeper waters (Fig. 7). Note, however, that these studies were conducted during spring and summer months and were limited almost

Table 1. Summary of Hawaiian monk seal diving studies.

Study	DeLong et al. (1984)	Schlexer (1984)	NMFS (unpubl. data)
Site	Lisianski	Lisianski	Laysan
Dates	4 May - 11 June 1980	1 July - 14 September 1982	29 March - 13 May 1992
Animals	6 adult males	4 adult males 1 sub female 1 juv female 1 juv male	11 adult males
Total dives	4,817	16,815	29,756
Total days	94	174	246
Dives per day	51	102	121
Maximum depth (m)	121	150-180	280
% Dives ≤ 40 m	59%	93%	63%

exclusively to adult males; diving behavior may vary by season, location, and age/sex class.

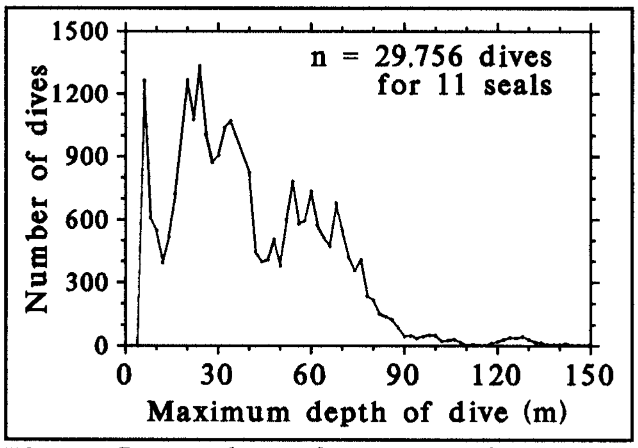

Figure 7. Number of dives with maximum depth of 10 to 150 m. Data are from 11 adult males instrumented with time-depth recorders at Laysan Island in March-May, 1992. Depths were measured in 2-m increments.

In September 1992, satellite transmitters were attached to 3 subadult males at French Frigate Shoals to investigate the location and depth of their foraging activities. Preliminary analyses indicate that the tagged animals spent a large portion of their time near the northern edge of the atoll (near the vicinity where they were tagged), and that the majority of their dives were less than 75 m maximum depth. (Results of this study will be reported after further analyses.) At present, satellite-linked time-depth transmitters appear to offer the most potential for obtaining data on foraging patterns because they provide information on surface location as well as depth. However, the transmitters used in the above study are large and will require some modification before they can be used extensively on juveniles. In spite of the considerable effort that will be required to reveal monk seal foraging ecology, with potential variation by age and sex class, season, and island population, the study of foraging habits is clearly important to the recovery and conservation of this species.

POPULATION AND STOCK STRUCTURE

Biological Basis of Populations

As noted above, the majority of Hawaiian monk seals are distributed among 5 primary populations at French Frigate Shoals, Laysan Island, Lisianski Island, Pearl and Hermes Reef, and Kure Atoll; smaller, less productive populations occur at Midway Islands, and at Necker, Nihoa, and Niihau Islands (Fig. 1). The relationships or "structure" among these populations can be evaluated by examining their correspondence (or, conversely, their variation) with respect to geographic distribution, life history and demographic parameters, characteristic phenotype, and characteristic genotype (Dizon et al. 1992).

The geographic distribution of the Hawaiian monk seal is difficult to evaluate because its pelagic behavior or ecology is poorly understood. The extent to which seals from different islands overlap in foraging distribution and at-sea movements is undetermined. However, as noted earlier, as much as 10 percent of the seals being recruited into the reproductive age classes (ages 6-10) are seen at sites other than their natal island. Such geographic dispersal argues that Hawaiian monk seals constitute a single reproductive unit.

On the other hand, demographic and life history parameters vary between island populations, suggesting a degree of independence. At French Frigate Shoals, for example, preliminary information on age of first birth, recent changes (1990-92) in birth rates and juvenile survival rates, growth rates, and physical condition are all at variance with other locations. Similarly, growth in abundance at French Frigate Shoals since the late 1950s is in marked contrast to the concomitant decline of the western populations. But such variation in parameters does not necessarily reflect inherent differences. The more recent changes at French Frigate Shoals may simply indicate those seals are environmentally stressed, which is consistent with the hypothesis that the number of seals at that site has approached K. In fact, during the 1990 reproductive season, birth rates at all locations dropped simultaneously, probably in response to a large-scale environmental phenomenon. The following year birth rates returned to more normal levels at other locations, but remained low at French Frigate Shoals where, again, demographic parameters may be more sensitive to environmental conditions. The longer term trends in monk seal distribution have not been fully explained, but evidence suggests that at Midway and Kure Atolls, as well as French Frigate Shoals, changes may have been related to varying levels of local disturbance from human activities (Kenyon 1972, Gerrodette and Gilmartin 1990).

Potential physical or phenotypic differences in monk seals from the different islands have not been systematically investigated. Size of immature animals at French Frigate Shoals is, at present, the only observed physical difference and, here again, the observed variation in growth rate is probably due to lack of food available to younger animals rather than any inherent population difference.

Similarly, the information currently available is inadequate for evaluating potential genetic differences among the island populations. Genetic variation may have been diminished significantly in the 19th century, with the reduction in monk seal numbers by sealers. Because existing populations are small, genetic drift following the reduction may have been relatively rapid, but inter-island movements of seals would have counteracted the development of genetic differences among populations. While potential genetic differences cannot be evaluated at the present time, samples of DNA are being extracted from molted fur and other tissues to look for such differences.

Hence, currently available biological information does not provide a basis for definitively describing the stock structure of the Hawaiian monk seal. But existing differences, primarily in demographic parameters, may result solely from environmental variation, and any potential for genetic divergence is mitigated by the inter-island movement of seals.

Recommended Stocks for Management Purposes

In spite of the uncertainties with respect to the stock structure of Hawaiian monk seals, the appropriate management unit can be defined using biological information; i.e., total species abundance. The relevant question is whether this species should be managed as a single unit (the entire species) or as multiple separate island populations (or subsets of populations). The choice of management unit is more or less important depending on whether the status of the species is defined in relative terms (i.e., relative to some upper limit such as K or OSP) or absolute terms (i.e., total absolute abundance). Because the absolute abundance of Hawaiian monk seals will always be small (due to limited and fixed habitat), this species will always remain dangerously close to extinction. The past 30-year period and the past 3-year period both demonstrate that their numbers can decline rapidly in a manner that may be essentially beyond management's control. Hence, the management unit should be chosen to provide the highest level of protection.

Given its current status, the most cautious choice may be to manage the Hawaiian monk seal as a single unit. The alternative, separate management of island populations, would increase the risk to the species if, for example, less protection was given to seals at French Frigate Shoals because their abundance is considered to be near local K. Importantly, the designation of the entire species as the management unit does not preclude management aimed at problems specific to individual populations.

POPULATION SIZE

Estimation Methods

Total abundance of Hawaiian monk seals is currently estimated by a combination of two principal methods, including (1) identification and enumeration of all individuals at a given site and (2) estimation of the number of seals at a site from beach counts and observed (known) haulout fractions (Gilmartin, et al. in press).

The first method, identification and enumeration of all individuals at a site, is clearly the more reliable method for determining abundance, but requires extensive effort and has not been possible during most field seasons at most sites. This method is much more difficult at larger multi-islet atolls (i.e., French Frigate Shoals and Pearl and Hermes Reef) where access to the islets is limited by weather and sea conditions, and where the process of identifying all animals can be highly disturbing because of the small size of the islets. Full identification is more feasible at single islands or small atolls with relatively large islets (i.e., Laysan and Lisianski Islands, Kure Atoll, and Midway Islands). Seals are identified on the basis of natural features such as scars or pelage marks, or by applied flipper tags or bleach marks. Bleach marks provide a temporary means of identification, as they last only until the seal molts, but are useful because many seals do not have obvious permanent marks and most older seals have not been tagged. Marking of all unidentifiable animals at an island requires a minimum of 4 weeks of effort, and this method therefore assumes that all unidentified seals seen early in the season remain at that location and are eventually identified or marked. Given this assumption, virtually all animals were identified at Lisianski Island in 1982-83 and 1992, Laysan Island in 1983-85 and 1988-92, and Kure Atoll in 1985, 1990, and 1992.

The second method of abundance estimation is based on known haulout fractions and beach counts (Gilmartin et al. in press). At established field stations, regular standardized beach counts (Stone 1984) have been conducted to determine, among other things, the number of hauled-out seals of each size and sex class (c_{ij}; where i and j index the various size and sex classes). These counts represent only a fraction (f_{ij}) of the local population because, at any given time, a portion of the seals are at sea. With a set of known haulout fractions, the abundance of each age class can be estimated using $\hat{N}_{ij} = c_{ij}/f_{ij}$, and the total number is $\hat{N}_{total} = \Sigma_i \Sigma_j \hat{N}_{ij}$. Abundance of pups is generally known and does not need to be estimated.

The required haulout fractions must be derived from records for years and locations where all individuals were identified; that is, this method of abundance estimation is dependent on information from the first method involving identification and enumeration of all seals at a site. Due to logistical constraints, the data for determining the f_{ij} are not available for certain sites (Nihoa Island, Necker Island, French Frigate Shoals, Pearl and Hermes Reef, and Midway Islands) and are only available for the remaining locations (Laysan and Lisianski Islands and Kure Atoll) in certain years. As Gilmartin et al. (in press) demonstrate, the haulout fractions vary seasonally and tend to be island- and year-specific; that is, they appear to be less reliable when used at sites other than where they were determined and when the time difference between the corrected beach counts and the haulout fractions is large (more than a few years). In spite of the difficulties inherent in these two methods, when combined they provide the best currently available means of estimating abundance of the Hawaiian monk seal. However, because large portions of each island population are marked, various mark-recapture techniques may provide more accurate and reliable estimates and should be investigated.

Population Estimates

The most recent estimates of abundance (Gilmartin et al., in press), ranged from 1,501 seals in 1984 to 1,976 seals in 1986, and then decreased to 1,752 seals in 1988. Totals from this assessment and for 1992 are listed in Table 2.

In 1992, all seals at Laysan and Lisianski Islands were identified; totals were 258 and 217 seals, respectively. Eighty-four seals were identified at Kure Atoll, but a small number of

Table 2. Estimates of the total Hawaiian monk seal population (including pups) for 1983 to 1988 (Gilmartin et al., in press), and for 1992. The upper number in each pair is the estimate and the lower number is the associated standard error. Note that estimates for Laysan Island in 1983 and 1985, and for Kure Atoll in 1985 are based on the method using haulout fractions and beach counts, but the populations were almost fully identified at those sites in those years. The estimates based on identification were lower (244, 295, and 68 for Laysan Island in 1983 and 1985 and Kure Atoll in 1985, respectively), and are probably more reliable.

Island	1983	1984	1985	1986	1987	1988	1992
French Frigate	820	714	826	919	885	850	654
Shoals	56	65	112	71	107	84	90
Laysan Island	280	269	331	383	442	331	258
	41		33	52	70		
Lisianski Island	253	215	234	309	266	242	217
		29	36	51	35	22	
Pearl and Hermes	78	111	115	173	122	129	228
Reef	14	20	17	40	18	14	34
Kure Atoll	64	58	73	60	64	67	92
	14	12	10	8	12	12	
Midway, Necker, and	131	131	131	131	131	131	131
Nihoa	23	23	23	23	23	23	23
Total	1,627	1,501	1,710	1,976	1,912	1,752	1,580
	76	78	126	112	136	92	147

adult-sized animals (6-10; H. Swensen, pers. comm.) could not be positively identified on the basis of individual markings and tags. Hence, the best estimate for the Kure Atoll population was approximately 90-94 seals. For the remaining populations, abundance was less certain.

At French Frigate Shoals in 1992, all pups (102) and juveniles (66) were identified, as well as 150 subadults and 306 adults. The individual identities of some subadults and adults were not determined and, therefore, the identified seals represent an unknown fraction of those size classes. Applying haulout fractions from 1992 Lisianski records[2] to beach counts from French Frigate Shoals (the method of Gilmartin et al., in press), the adult total was estimated as 336 (standard deviation [sd] ± 78.6) and the subadult estimate was 124 (sd ± 44.2). The estimate of adults, 336, was considered the best available. But this method underestimated the number of subadults, and 150 (the number identified) was used as the best estimate. The total for all four size classes, then, is 654 seals. Comparison with the 1988 total estimate (850), indicates a decline of 23 percent in the total number of monk seals at French Frigate Shoals between 1988 and 1992. During this 4-year period, the mean beach counts at this site declined by approximately 27 percent. The determination of a variance for the 1992 total estimate was problematic unless the variance estimate for subadults from the method based on haulout fractions was used as the best available, in spite of the fact that the method underestimated the number of seals in this size class. Assuming zero variance in the estimates of pups and juveniles, the sum of the variances for subadults and adults indicated a standard deviation for the total estimate on the order of 90 seals.

The estimate of 1992 abundance at Pearl and Hermes Reef was also questionable because only 5 censuses were conducted over a 5-day period from 20-24 July. Using the method of Gilmartin et al. (in press) and the 1992 haulout fractions from Lisianski (for July only), the estimates for adults, subadults, and juveniles were 103, 42, and 57 respectively (combined sd ± 34.3). Twenty-six pups were tagged during July and, using 26 as the best estimate for the number of pups born, the total estimate for Pearl and Hermes Reef was 228 seals. This is well above the 1991 estimate of 160-170, which was based on an effort to bleach mark all seals (Finn et al. in prep).

Gilmartin et al. (in press) estimated the combined population at Necker, Nihoa, and Midway Islands as 131 (sd ± 23).

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[2] In estimating abundance of the French Frigate Shoals population between 1983-88, Gilmartin et al. (in press) used haulout fractions from Laysan Island. Haulout fractions from Lisianski Island were used here because 1992 field records from Laysan were not yet available.

Seals at Necker and Nihoa Islands have not been monitored routinely and, lacking evidence of change at those sites, the estimate by Gilmartin et al. (in press) still may be the best available. The population at Niihau Island is unstudied, but probably relatively trivial (in a numerical sense) and therefore not included here.

Minimum Population Size

For the majority of island populations, minimum abundance estimates were based on the number of seals identified. At Laysan and Lisianski Islands, the minimum estimates were the same as the best estimates of 258 and 217 seals, respectively. Eighty-four seals were identified at Kure Atoll (which is only slightly below the best estimate of 90-94), and at French Frigate Shoals, 624 were identified. Virtually no efforts were made to identify seals at Pearl and Hermes Reef; hence, the best minimum estimate for this population (based on the best estimate minus 2*sd) was 228 - 2 * 34 = 160 seals. Similarly, the minimum estimate for Necker, Nihoa, and Midway Islands combined was 131 - 2 * 23 = 85 seals. The total minimum estimate for all island populations was 1,428.

POPULATION GROWTH RATES AND TRENDS

Trends for the Species

Total abundance is not estimated annually for the Hawaiian monk seal. Instead, two primary indices are used for assessing trends in the status of the species and the individual populations. These indices are annual number of births and mean annual beach counts (Fig. 8). Because a large portion of the species resides at French Frigate Shoals, changes in births and beach counts at that site have a correspondingly large effect on these indices. Since 1983, the total number of births has varied considerably, increasing from 167 to 226 between 1985-88, declining sharply to 142 in 1990, and increasing again to 209 in 1992. Such annual variability obscures longer term trends and suggests that reproduction in Hawaiian monk seals is a relatively unstable population parameter. This lack of stability is an important consideration in projecting future trends for this species. The total of mean beach counts for the 5 main populations appears to be less variable, but shows a general decline since 1986 (Fig. 8, bottom).

Birth rates and other demographic parameters may be highly correlated, as was seen in 1990 when the number of births dropped at all locations. However, in addition to range-wide trends, the characteristics and dynamics of the different populations are known to vary and, for that reason, will be discussed separately.

Trends by Population

French Frigate Shoals. Contrary to the decline of the western populations after the late 1950s, the number of monk seals at French Frigate Shoals grew to a level thought to be near its local K (Fig. 9). By the mid-to-late 1970s, growth appeared to be slowing, but the mean beach count (including pups) rose substantially in 1985, remained high through 1989, and then declined sharply in 1990-92. Approximately 46-50 percent of all Hawaiian monk seals were located at French Frigate Shoals in 1983-88; in 1992, 42 percent were at this atoll.

Gerrodette and Gilmartin (1990) attributed the growth of this population after the late 1950s to a decrease in disturbance at East Island, the primary pupping site at French Frigate Shoals. A U.S. Coast Guard loran station was located at East Island between 1944 and 1952, and was then moved to Tern Island. The removal of disturbance from the primary pupping site probably lead to increased pupping, survival of pups and juveniles, and subsequent growth in the number of seals at this atoll.

Coming after three decades of growth, the decline in the number of seals since 1990 has been a matter of substantial concern. This decline is reflected in mean beach counts and number of births, as well as birth rate, survival of juveniles (Fig. 10), and condition of

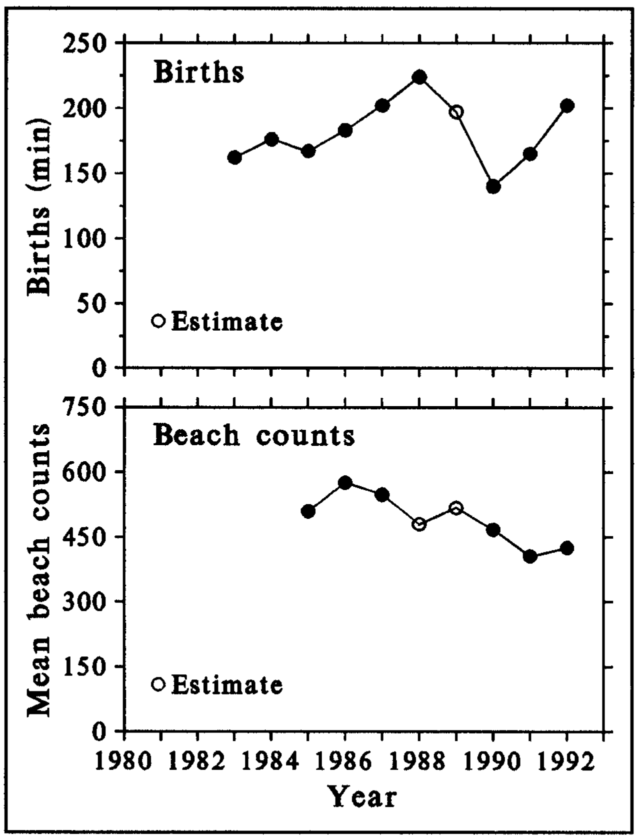

Figure 8. Minimum number of annual births and mean beach counts from the 5 main populations of Hawaiian monk seals, 1982–92.

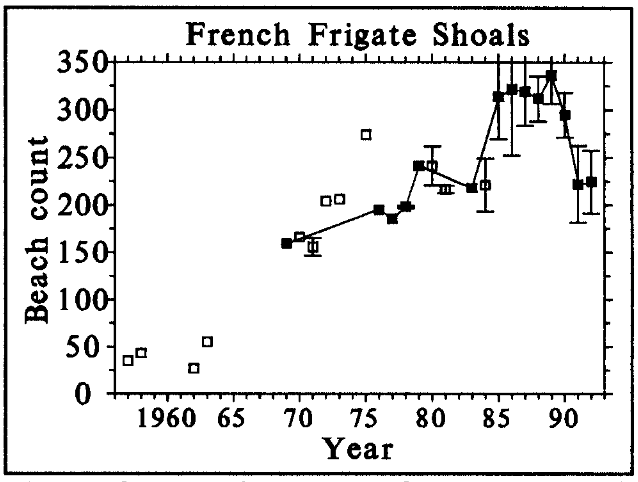

Figure 9. Beach counts of Hawaiian monk seals at French Frigate Shoals, 1956–92. See Appendix B for data and sources used to reconstruct the trend.

surviving juveniles (Fig. 6). The number of births (Fig. 10a) in 1991 was the lowest recorded since intensive monitoring began in 1984. The apparent cause was a decrease in birth rate of mature females rather than a loss of females (Fig. 10b). The birth rate of identified adult females dropped precipitously in 1990 (0.33, n = 97), remained low in 1991 (0.36, n = 158), and recovered, at least partially, in 1992 (0.51, n = 118).

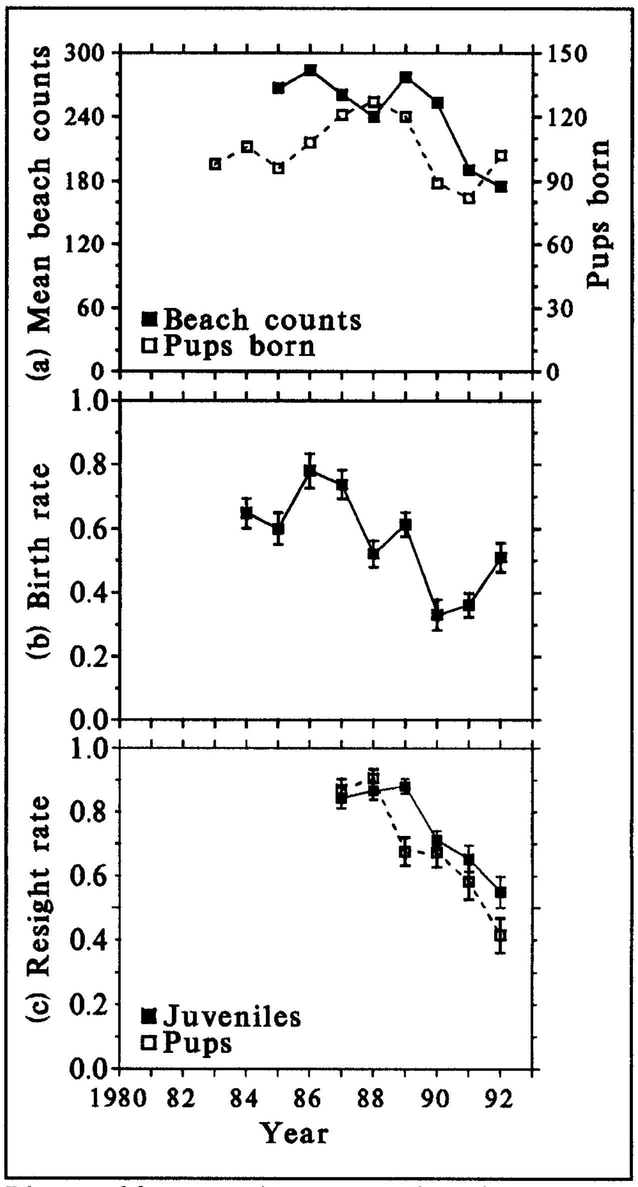

Figure 10. a) Mean beach counts (excluding pups) and number of pups born, b) birth rate of identified adult females, and c) resight rate of juveniles and pups at French Frigate Shoals.

Survival of seals born during this period was also poor. Figure 10c illustrates the observed decline in resighting rate for immature seals after 1988. Resighting effort during this period has been extensive, and the probability of sighting was at least as good as in previous years. Also, large numbers of these animals were not observed at other locations, indicating that the drop in resighting cannot be explained by emigration. Note, however that Necker and Nihoa Islands were not regularly monitored, and that seals from French Frigate Shoals have been sighted at those locations. Movements from French Frigate Shoals to these islands have not been quantified, but the full extent of those movements was probably much less than required to explain the losses at French Frigate Shoals. Hence, the drop in resighting rates was assumed to indicate a corresponding drop in survival. Survival rates from birth through mid September 1992 were especially poor for the last 4 cohorts (Table 3).

Also, as noted earlier, many surviving juveniles have been in poor condition which, presumably, compromises their survival

Table 3. Composition and survival of Hawaiian monk seal cohorts born in 1984-92. Survival to 9/92 was estimated as a range. The lower end of the range was calculated by assuming that all the removed seals would have died; hence, this estimate is biased downward. The upper end of the range was calculated by ignoring the removed seals (i.e., number surviving/[number tagged-number removed]), and is probably biased upward because the expected survival of the removed seals (had they been left at French Frigate Shoals) was less than the expected survival of the seals not removed. Approximate annual survival calculated as $(l_x)^{(1/age)}$.

Tag year	Age	Sex	Number tagged	Number removed	Number surviving	Survival to 9/92 (l_x)	Approximate annual survival
1984	8	F	36	4	27	0.75–0.84	0.96–0.98
		M	42	3	21[a]	0.50–0.54	0.92–0.93
1985	7	F	38	2	19	0.50–0.53	0.91–0.91
		M	48	–	17	0.35–0.35	0.86–0.86
1986	6	F	48	5	24	0.50–0.56	0.89–0.91
		M	52	1	17	0.33–0.33	0.83–0.83
1987	5	F	51	–	21	0.41–0.41	0.84–0.84
		M	55	–	25	0.45–0.45	0.85–0.85
1988	4	F	62	8	15[b]	0.24–0.28	0.70–0.73
		M	52	–	15	0.29–0.29	0.73–0.73
1989	3	F	50	3	16	0.32–0.34	0.68–0.70
		M	51	–	17	0.33–0.33	0.69–0.69
1990	2	F	38	11	6	0.16–0.22	0.40–0.47
		M	41	–	12	0.29–0.29	0.54–0.54
1991	1	F	44	6	30	0.68–0.79	0.68–0.79
		M	24	–	6	0.25–0.25	0.25–0.25
1992	0	F	49	15	–	–	–
		M	35	–	–	–	–

[a] Includes 3 males sighted at Laysan Island in 1992.
[b] Includes 1 female sighted at Nihoa Island in 1992.

to maturity. The slower growth rate and relatively poor condition of juvenile females could result in an older age of first birth at French Frigate Shoals, as indicated in Figure 3.

Possible explanations for these trends at French Frigate Shoals include (1) increased disease or poisoning (e.g., ciguatoxin), (2) increased shark predation, (3) increased injuries inflicted by other seals, (4) increased disturbance at hauling areas, (5) increased emigration, (6) changed hauling

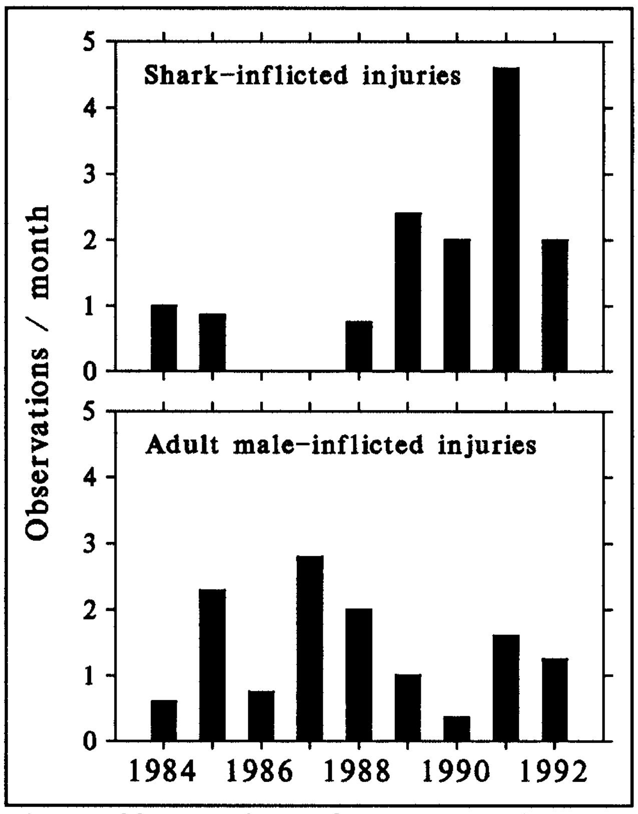

Figure 11. Number of injuries observed per month, inflicted by sharks (gaping wounds and amputations; top) and adult males (gaping dorsal wounds; bottom) at French Frigate Shoals, 1984-92.

patterns with more time spent at sea, or (7) decreased prey availability due to predation by the large number of seals (i.e., the population is near K and has reduced its own food supply), competing fisheries, or some environmental phenomenon not directly related to monk seals and fisheries.

Several of these possibilities can be ruled out as the primary cause for the decline. Disease studies were conducted in April-May 1992 (Gilmartin and Ragen 1992), and again in September 1992, and included blood cell counts, blood chemistries, specific tests for distemper (canine and phocine), caliciviruses, parvoviruses, leptospirosis, salmonellosis, and gastrointestinal parasites. Varying levels of parasitic infestation were found, but no significant disease processes were revealed. Histopathology and general necropsy findings from dead animals also failed to reveal a significant disease problem.

Similarly, ciguatoxin poisoning has probably not increased in prevalence at French Frigate Shoals during the past 3 years. The expected signs of ciguatoxin poisoning include a range of gastrointestinal, neurological, and cardiovascular disorders that occur over a period of days to weeks or longer (Withers 1982). If the die-off of animals at Laysan Island in 1978 was due to ciguatoxin, the observed signs of disease should have been detectable at French Frigate Shoals had a large-scale poisoning occurred.

Injuries and deaths are known to occur from attacks by sharks. Injury reports from 1984-92 suggest that the number of severe injuries (defined here as gaping wounds and amputations) attributed to shark predation increased substantially after 1987-88 (Fig. 11). Shark predation could account for the observed decrease in survival and may continue at a high level as long as large numbers of seals at this site are in poor condition. But such predation is less likely to have caused the low birth rate of adult females and poor condition of juvenile seals. Hence, the apparent increase in shark predation may be related to the decline of monk seals at French Frigate Shoals, but is probably not the underlying cause.

Injuries and death also result from mobbing by multiple adult males and similar aggression by single males. Mobbing injuries are less likely to be observed at French Frigate Shoals because the atoll is larger and is not monitored with the same regularity and frequency as a single island such as Laysan Island. However, the number of recorded injuries (Fig. 11) does not suggest an increase in adult male aggression that could account for the observed decline. As was just mentioned for shark predation, adult male aggression might account for some portion of the observed mortality of juvenile seals, but does not explain the decline in the condition of juvenile seals or the birth rate of adult females.

Disturbance at French Frigate Shoals has been at a relatively low level since the loran station at Tern Island closed in 1979. Hence, the decline over the past 3 years is probably not due to trends in human activity. As noted earlier, large-scale emigration of seals from French Frigate Shoals has not been observed during annual monitoring of most of the remaining islands and atolls where seals haul out. Changes in seal hauling patterns (i.e., with more time at sea) could explain lower beach counts, and might be consistent with the observed poor condition of juveniles. However, given the extensive resighting effort, changes in hauling patterns would not account for the low number of seals resighted at least once during a season. In effect, hauling patterns are more likely a consequence, rather than a cause, of the underlying problem.

Finally, the observed decline at French Frigate Shoals may be due to decreased prey availability. This hypothesis is consistent with all of the observed changes in reproduction, survival, and condition of surviving immature animals. An evaluation of this hypothesis requires knowledge of the principal monk seal prey and its abundance or availability. These prey may have declined from excessive predation by monk seals, overfishing, natural processes unrelated to monk seal abundance or fisheries, or some combination of these factors.

The type and relative importance of monk seal prey are still largely unknown. As mentioned earlier, known monk seal prey include shallow benthic and coral reef fishes, octopus, lobster, and some deeper benthic prey. However, some prey items may not be found in scats and spews and have therefore gone undetected. The relative importance of nearshore versus pelagic prey has not been evaluated. Furthermore, the available evidence from scats and spews is, as yet, insufficient to determine if prey preferences differ among island populations, by age and sex class, or by season of the year. To address the question of whether or not prey have declined, present efforts are forced to evaluate broad categories of potential prey with the assumption that monk seals are opportunistic, catholic feeders.

In July 1992, NMFS assessed potential prey at French Frigate Shoals. At selected sites fish abundance was surveyed for comparison with a similar study in 1980-83. Results indicated that the total number of fishes had not declined over the past decade, but that the composition had shifted, with fewer secondary consumers (benthic and zooplanktivorous fishes). The number of herbivorous fish did not decline, but they tended to be distributed more around patch reefs in the interior of the atoll rather than at the fringing reef, where they were found in the earlier surveys. At the present, these results are difficult to relate to the monk seal decline because, again, the relative importance of these fishes, particularly as prey for weaned pups and juveniles, is not understood.

As the principal monk seal prey are not yet identified, causes for a hypothetical decrease in prey availability are also difficult to assess. The observed changes in monk seal parameters are all consistent with the possibility that the number of seals has increased to the extent that they have simply reduced their prey to some critical level or range. If monk seals are primarily nearshore feeders, the distribution of their prey may be very limited and fixed, and the increase in seals observed at French Frigate between the 1950s and 1970s may have been sufficient to fully exploit the available prey base.

At the same time, the available prey may have been reduced by fisheries. Consider, for example, the lobster fishery. Catch-per-unit-effort declined for the lobster fishery by almost 80 percent between 1982 and 1990 (Landgraf 1991). In 1990, fishing effort in the NWHI was the second highest recorded, but the landings (in weight) were the lowest recorded since 1983 (Landgraf 1991). Furthermore, 50 percent of the lobsters taken were illegal; i.e., either too small or berried (see Landgraf 1991, his Table 7). A relatively small percent of the total number of lobsters taken in 1990 in the NWHI were taken at French Frigate Shoals, but almost 90 percent of the catch was taken near Necker Island or Gardner Pinnacles, approximately 150 and 200 km from French Frigate Shoals, respectively. Monk seals are seen at both of these sites and they may be important feeding areas for seals from French Frigate Shoals.

Similarly, the total bottomfish catch per fishing day in the NWHI declined each year from a mean of 473 kg in 1987 to 268 kg in 1991, a decrease of 43 percent (Kawamoto 1992). The importance of bottomfish in the monk seal diet is unknown, so the significance of changes in this fishery cannot yet be determined.

Pelagic fisheries, primarily longlining, are not likely to compete substantially with monk seals for prey; instead, the relationship between these fisheries and monk seals is more likely to involve direct interaction of seals with catch and gear. Monk seals are known to interact with these fisheries, but the available data are not sufficiently reliable to assess quantitatively the extent of the interactions. For example, Dollar (1992) reported, ". . . out of 1,665 longline trips completed in 1991, only 118 trips (7%) reportedly had any type of interactions with protected species, whereas data from the 10 observer trips indicated 60 percent of the trips had interactions."

Observed changes in both monk seal population parameters and fisheries also are consistent with a third possible explanation for a decline in prey availability; i.e., a shift in marine conditions in the NWHI that resulted in lower productivity of one or more key prey species throughout the ecosystem. In December 1992, NMFS and the Joint Institute for Marine and Atmospheric Research of the University of Hawaii sponsored a "Workshop on Variation in the Marine Environment and Ecosystem around the Hawaiian Archipelago." Oceanographic data presented at the workshop indicate that from the late 1970s to the late 1980s wind-driven surface mixing was greater, leading to a deepening of the mixed layer. Greater mixing may have supported increased levels of productivity around the Hawaiian Archipelago. Observed biological changes in the late 1980s and early 1990s may therefore reflect a return to more characteristic oceanographic conditions. Data on seabirds, lobsters, monk seals, and coral reef fishes were also presented at the workshop, and were consistent with the overall hypothesis of a decadal scale shift in oceanographic conditions and productivity.

Regardless of the reason(s) for decreased prey availability, the situation at French Frigate Shoals will require continued, extensive monitoring and management efforts. For the reasons just described, it is not clear that the observed changes simply represent natural demographic regulation which will eventually result in a stable population. Even if these changes reflect natural processes, the relatively high mortality of juveniles is a significant loss to the species. If young seals that are likely to perish can be captured and rehabilitated, then they can be released at other sites where populations have not yet recovered from the declines of the past several decades. The successful movement of young females from French Frigate Shoals to Kure Atoll demonstrates that such translocations are an effective method of enhancing the recovery of the Hawaiian monk seal.

Laysan Island. The Hawaiian monk seal population at Laysan Island declined significantly between the late 1950s and the early 1980s (Fig. 12); much of this decline is unexplained.

Beach counts varied widely from the late 1950s through the mid 1970s; the observed variation in this period cannot be parsed into natural variation in number of animals hauled out and measurement error, such as might result from disturbance of animals and disruption of their normal hauling patterns (Kridler 1964, Ely and Clapp 1973). Many of the values included in Figure 12 for years prior to the late 1970s were derived from single counts. More frequent, standardized counts were conducted beginning in the late 1970s (DeLong 1976; DeLong et al., 1976; Johnson and Johnson 1978, 1980, 1981a, 1981b, 1984). The decreased variation in the more recent counts suggests that much of the earlier variation might be attributed to measurement error (primarily from nonstandardized methods and the use of single counts or count means where very few counts occurred). Thus, while the earlier counts indicate that a substantial decline occurred, the nature of that decline is difficult to investigate. In contrast, at least some of the more recent variation can be explained. For example, the trend indicates a substantial decline in 1978, when a die-off was observed on Laysan Island (Johnson and Johnson 1981a). However, in spite of intensive monitoring, the cause of the apparent increase and then decrease in the late 1980s is not understood. Since the late 1980s, beach counts have remained relatively stable. The population also appears to have recovered from the sharp decline in number of births and birth rate in 1990 (Fig. 13). Thirty-eight pups were born in 1992, which is the second highest number of births recorded at this site in the last 12 years of intensive monitoring.

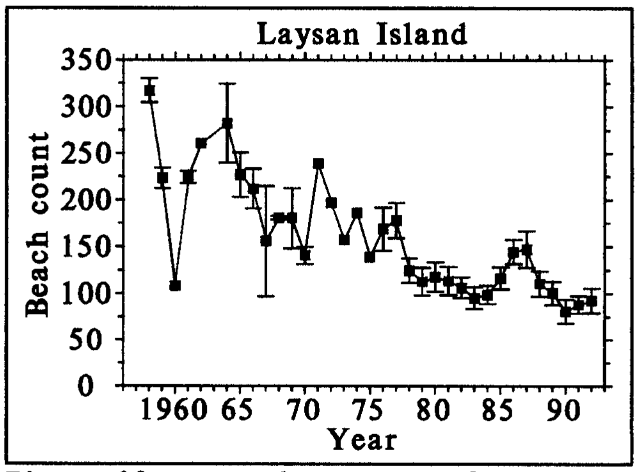

Figure 12. Beach counts of Hawaiian monk seals at Laysan Island, 1956–92. See Appendix B for data and sources used to reconstruct the trend.

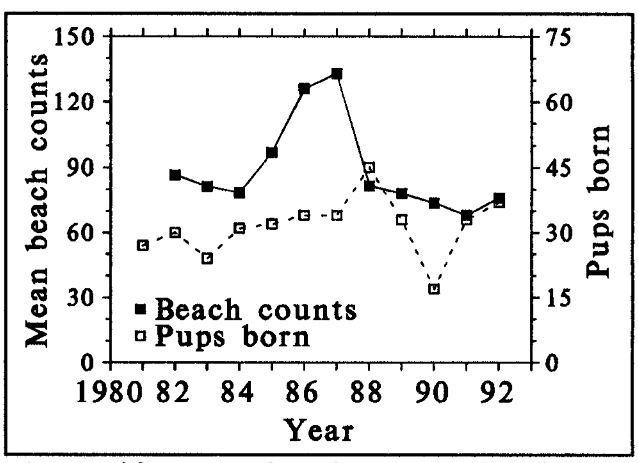

Figure 13. Mean beach counts (excluding pups) and number of pups born at Laysan Island, 1981–92.

Beginning in 1983, the composition of the Laysan population has been determined each year except 1986-87 (Fig. 14). As illustrated, the number of adult males decreased slightly over the last 5 years, whereas the number of adult females remained approximately stable. The numbers of subadults and juveniles dropped by approximately half between 1988 and 1991; subadults increased slightly in 1992, whereas juveniles increased by about 60 percent.

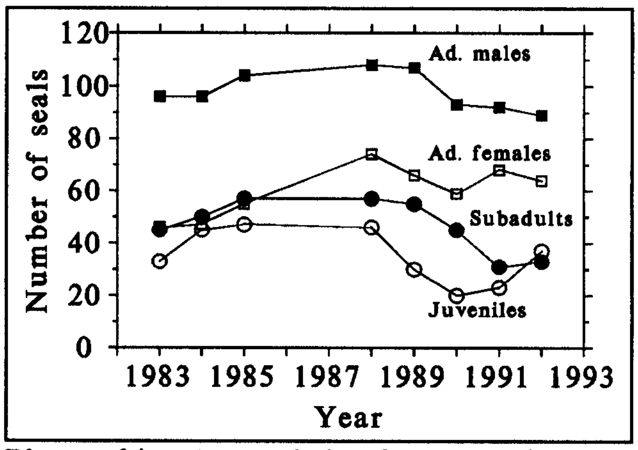

Figure 14. Composition by size class of the Laysan Island population, 1983-92.

The composition of the population at Laysan Island has been a concern because the adult sex ratio has been heavily skewed toward males, and this imbalance has been considered a key factor in the occurrence of mobbing events at this site. As illustrated in Figure 15, the sex ratio of adults has been slowly returning to a more normal (or expected) level near 1:1. However, the decline in the sex ratio has not been accompanied by a decline in mobbing-related deaths and disappearances (Fig. 16). In 1992, 7 seals were known to die from mobbing (2 adult females, 1 subadult female, 1 adult male, 2 subadult males, and 1 juvenile male) and an additional 3 seals (1 adult female and 2 subadult females) were seen with serious mobbing injuries and then not seen again. These numbers include only observed victims; mobbing occurs in the water, and other mobbed seals may have died before returning to the island where they could be observed. Between 1982 and 1992, 25 dead adult females have been recorded; 22 (88%) of those were mobbed prior to death. Similarly, of 59 recorded deaths of seals older than nursing pups, 42 (71%) of those had been mobbed prior to death. Hence, mobbing has been and continues to be a major factor inhibiting the recovery of this population.

Management efforts to ameliorate mobbing have focused primarily on restoration of the adult sex ratio to a (presumably) normal level closer to 1:1. Removal of males has been considered the most feasible means of restoring the ratio. Removal can be achieved by a number of mechanisms. Lethal removal is irreversible, and in a practical sense, transfer to other locations may also be irreversible. Potential

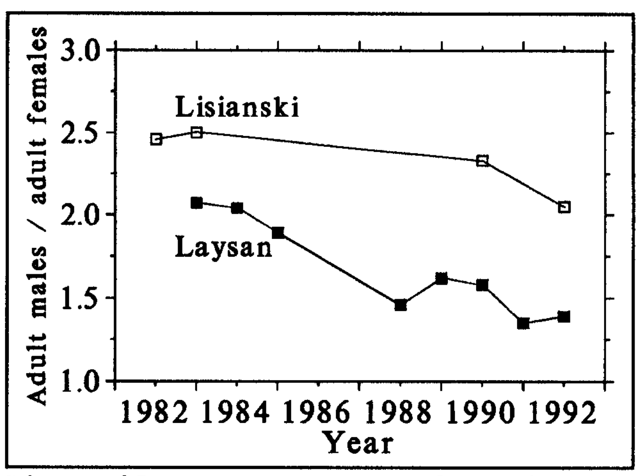

Figure 15. Sex ratio of adults at Laysan and Lisianski Islands, 1982-92.

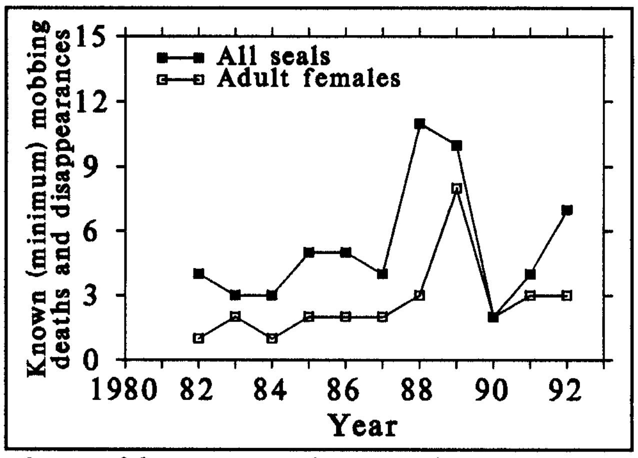

Figure 16. Known (minimum) deaths and disappearances of all seals (older than nursing pups) and adult females at Laysan Island, 1982-92.

problems with irreversible removal include (1) some degree of reproductive failure due to insufficient numbers of mature males, and (2) disruption of the male dominance hierarchy by removal of dominant males. Reproductive failure due to insufficient numbers of males is considered unlikely because estrus in adult females is asynchronous and a single male is able to mate with multiple females. Disruption of the dominance hierarchy may have serious consequences if dominant males are removed unintentionally and no subordinant males are able to establish dominance and thereby control access to females in estrus. The disruption may be relatively short-lived (correcting within a season) or long-lived (persisting over several seasons), particularly if future dominant males are removed. The plasticity of the hierarchy is unknown, but mobbing might increase in such situations, and the selection of males for removal is, potentially, very important.

Because these consequences are not reliably predictable (at the present time) a plan was developed (Gilmartin and Alcorn 1987) to attempt reversible removal using a testosterone-suppressing drug which has been successfully tested on captive seals (Atkinson et al. in press). A field trial was conducted in 1992, and Decapeptyl® was administered to 10 adult males. The treated males were chosen on the basis of behavior patterns in 1991 and behaviors of known mobbers from previous years. After treatment, these males (and 2 groups of controls) were observed on a daily basis to evaluate their behavior and potential responses to the drug. The 10 treated seals and 9 controls were also fitted with time-depth recorders to allow comparison of diving and hauling patterns. Blood samples were taken when the drug was administered, and 2-6 weeks later; both sets of samples were evaluated to measure the drug's effect on testosterone blood levels. Results are currently being evaluated, and will be used to determine the strategy for management efforts to facilitate the recovery of this population.

Lisianski Island. The Lisianski population also experienced an unexplained and severe decline after the late 1950s (Fig. 17). The apparent variability in beach counts at this site was much less, however, and the decline appears to have occurred prior to the mid 1970s. For the past 2 decades, beach counts have remained relatively low and stable. Counts increased in 1978-83, dropped sharply in 1984, rose again until 1986 and declined thereafter. The number of births, which has been monitored since 1982, has also been relatively variable (Fig. 18). Pup production was particularly low in 1984-85 and in 1990-91. Twenty-three pups were born in 1992.

The composition of the beach counts (Fig. 19) during the past decade suggests that the small amount of observed variability is due to fluctuation in the number of adults. In spite of this fluctuation, the sex ratio of adults has remained persistently and strongly in favor of males (Fig. 15). Mobbing is observed at Lisianski, but its incidence has been more difficult to quantify because monitoring has been less regular at this site than at Laysan Island. In 1992 one death was known to result from mobbing (subadult female), but 12 other seals were observed with severe dorsal injuries characteristic of wounds inflicted by adult males. Mobbing is considered a likely explanation for this population's failure to grow.

A second factor which may have contributed to the lack of growth at this site is entanglement in marine debris. Henderson (1990) reviewed entanglements of Hawaiian monk seals by location between 1982-88 and found the highest rate of entanglement at Lisianski Island (4.44 incidents observed per 100 field camp days per 100 seals observed). In 1992, 8 of 14 observed entanglements occurred at this site. In spite of these

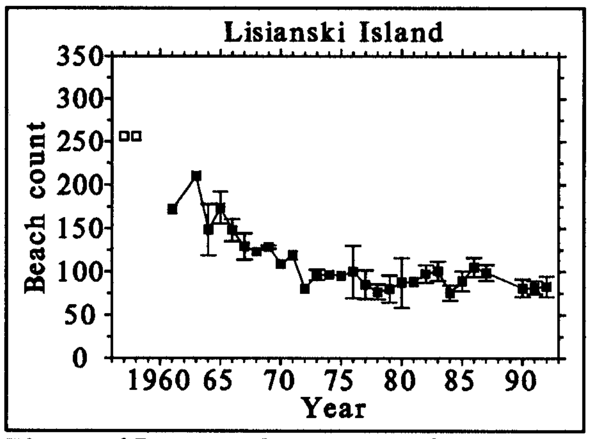

Figure 17. Beach counts of Hawaiian monk seals at Lisianski Island, 1956–92. See Appendix B for data and sources used to reconstruct the trend.

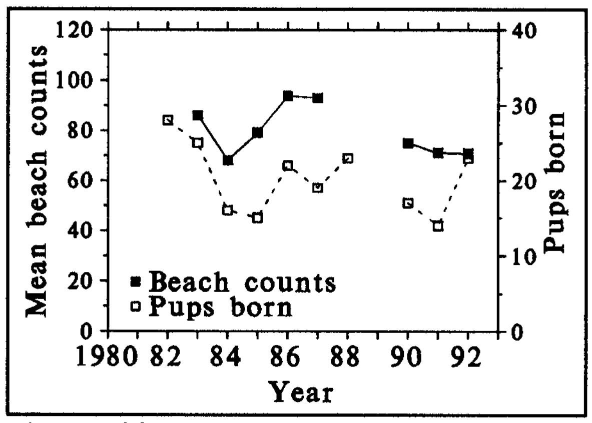

Figure 18. Mean beach counts (excluding pups) and number of pups born at Lisianski Island, 1982–92.

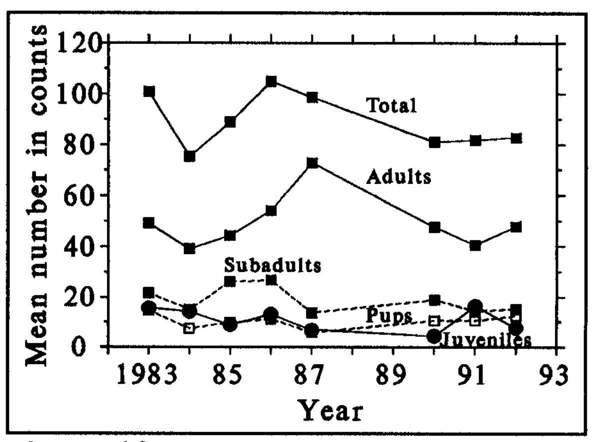

Figure 19. Composition by size class in beach counts at Lisianski Island, 1983–92.

observations, the significance of entanglement as an impediment to growth of this population is uncertain.

Pearl and Hermes Reef. The Pearl and Hermes Reef population also experienced a severe decline between the late 1950s and early 1970s (Fig. 20). However, since the recorded low in 1975, monk seals at Pearl and Hermes Reef appear to be making a slow steady recovery. The composition of this population has never been fully determined, but beach counts (Fig. 21) indicate growth of each size/sex class and the overall trend for the beach count total is positive.

The number of births at Pearl and Hermes also appears to be increasing (Fig. 22). The low number of births recorded for 1990 was consistent with the drop in number of births at all locations in that year. However, the 1990 field effort at this atoll was limited to 2 days and, in all likelihood, the number of births was underestimated. In 1992, only 5 censuses were conducted over a 5-day period in late July. Twenty-six pups were identified, which is the largest number of pups counted at this atoll since monitoring by NMFS began in 1983.

Monk seals at Pearl and Hermes Reef are generally considered to be in good condition. Mobbing is not known to be a problem, and the adult sex ratio appears to be approximately 1:1 (J. Henderson, pers. comm.). This population is expected to continue its recovery.

Kure Atoll. The Kure Atoll

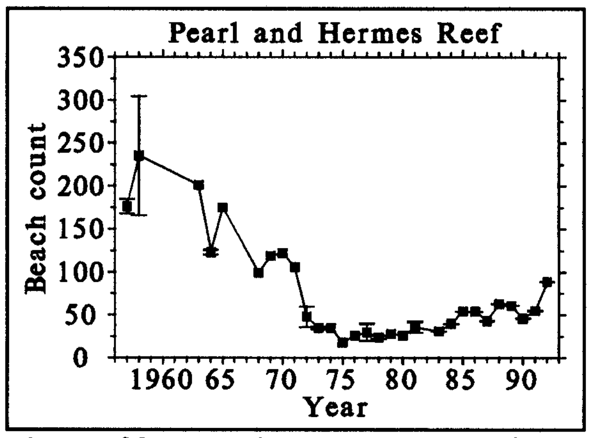

Figure 20. Beach counts at Pearl and Hermes Reef, 1956-92. See Appendix B for data and sources used to reconstruct the trend.

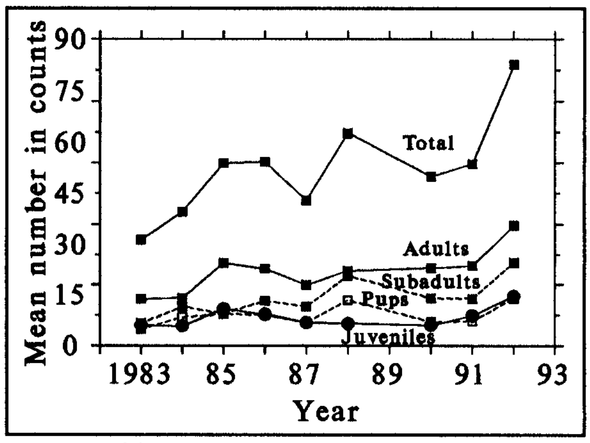

Figure 21. Composition by size class in beach counts at Pearl and Hermes Reef, 1983-92.

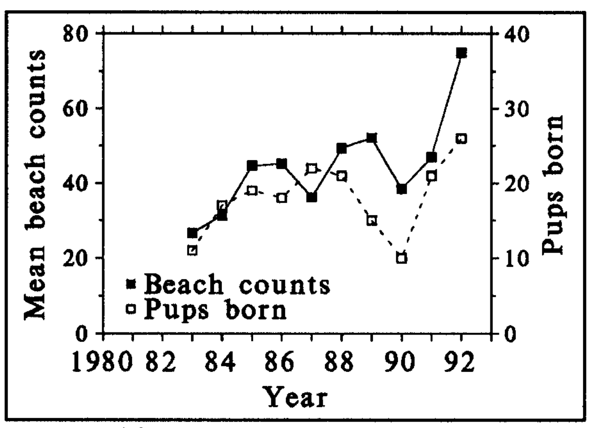

Figure 22. Mean beach counts (excluding pups) and number of pups born at Pearl and Hermes Reef, 1983-92.

population also appears to be recovering. Abundance at this atoll declined abruptly in the late 1950s and early 1960s following the construction and occupation of a U.S. Coast Guard loran station on Green Island (Fig. 23; see Gerrodette and Gilmartin 1990). As discussed earlier, Kenyon (1972) attributed this decline to abandonment of primary pupping areas due to human disturbance. Pup survival fell (Wirtz 1968), followed by failure of reproductive recruitment and the development of an imbalanced age structure skewed toward older animals. In addition, the sex ratio of adults became heavily skewed toward males (VanToorenburg et al., 1993). Births declined steadily from the late 1970s to the mid 1980s and only a single pup was born in 1986. Thereafter, the number of births increased steadily (again, with the exception of 1990, when births were low at all locations) to 14 in both 1991 and 1992 (Fig. 24).

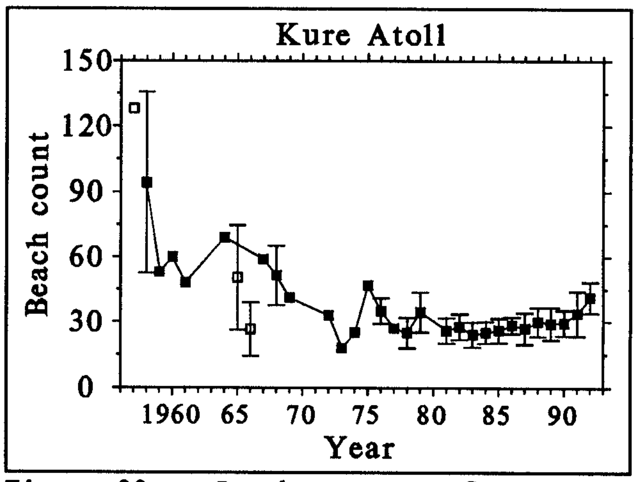

Figure 23. Beach counts of Hawaiian monk seals at Kure Atoll, 1956-92. See Appendix B for data and sources used to reconstruct the trend.

The general downward trend at Kure Atoll was reversed by modification of U.S. Coast Guard regulations to minimize disturbance of seals, particularly at the primary pupping locations, and by intensive management initiated by NMFS in 1981. The Head Start program, described earlier, protected weaned female pups from sharks and adult males and, indirectly, provided additional opportunity for biologists to observe and encourage U.S. Coast Guard efforts to minimize disturbance of seals. Thirty-three pups were included in this program between 1981-1991, and 24 of those were sighted in 1992. The rehabilitation program, started in 1984, added 20 yearling females to the population (of which 13 were sighted in 1992), and 5 additional yearling females were transported directly from French Frigate Shoals to Kure Atoll in 1990. Altogether, these restoration efforts

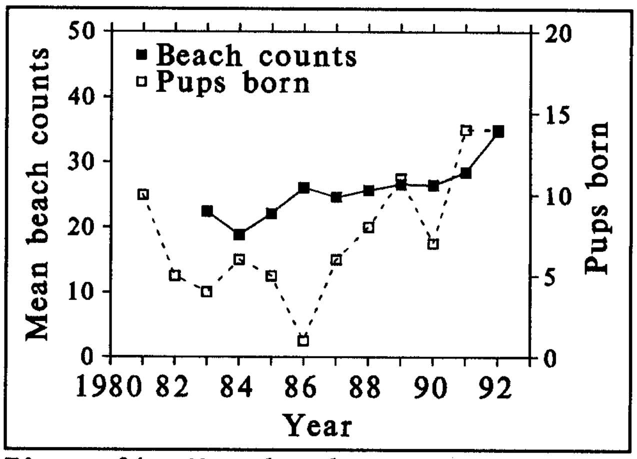

Figure 24. Mean beach counts (excluding pups) and number of pups born at Kure Atoll, 1981-92.

successfully augmented immature size classes, enhanced reproductive recruitment (Fig. 25), and returned the adult sex ratio to a presumably normal level (VanToorenburg et al., 1993).

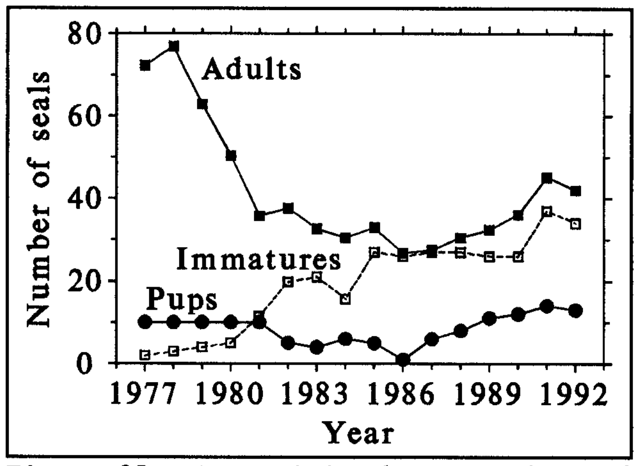

Figure 25. Composition by size class of the Kure Atoll population, 1977–92.

In July 1992, the U.S. Coast Guard closed and vacated the loran station on Green Island. The beach enclosure was removed and weaned female pups were not held in captivity as in previous years. In addition, rehabilitated female yearlings from French Frigate Shoals were taken to an enclosure at Midway Islands rather than Kure. In spite of the discontinuation of these programs at this site, the Kure Atoll population will continue to be monitored annually, and is expected to continue its recovery.

Midway Islands. In contrast, the Midway Islands population shows no signs of recovery. Historical records suggest that this population has been depleted at least since the late 1800s. The highest recorded counts occurred in 1957-58 (Kenyon 1972, 9 counts, $\bar{x}$ = 55.7, sd ± 9.1). But during those counts virtually all of the observed seals were limited to 3 small islets between Sand and Eastern Islands (the 2 main islands), suggesting that the population was well below its carrying capacity. Within a decade of those high counts, the population was almost extinct (Fig. 26). A single seal was observed during an aerial survey in March 1968 (Kenyon 1972) and, thereafter, this population has failed to recover.

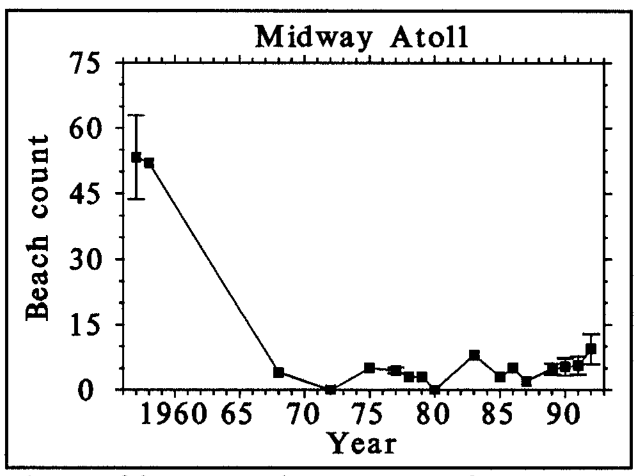

Figure 26. Beach counts of Hawaiian monk seals at Midway Islands, 1956–92. See Appendix B for data and sources used to reconstruct the trend.

As at Kure Atoll, the decline of the Midway population has been attributed primarily to human disturbance and its effects on reproduction and juvenile survival (Kenyon and Rice 1959, Rice 1964, Kenyon 1972). Permanent human habitation of the atoll began in 1902, with the establishment of a cable station. A runway was built in 1935 by Pan American

Airways, and the atoll was exposed to severe disturbance during World War II (Bryan 1956). The number of human inhabitants declined in the late 1940s but increased again in the 1950s to a maximum of nearly 3,000. Kenyon (1972) reported that in the late 1950s access to Eastern Island was restricted to island personnel and the small islets between Eastern and Sand Islands were rarely visited. In 1968, access to these areas was not restricted and visitors were more frequent. In 1978 the human population, which had decreased to about 1,600, was further reduced to approximately 250 and has subsequently remained at that level.

In spite of the reduction in the human population at Midway, the Hawaiian monk seal population has remained on the verge of extinction. Kenyon (1972) suggested that the number of seals at this atoll should be comparable to populations at Pearl and Hermes Reef, and Laysan and Lisianski Islands. Hence, in contrast with other small populations at Necker and Nihoa Islands, the population at Midway should have significant potential for growth. Because of this potential, and because the U.S. Coast Guard loran station at Kure Atoll has been closed, rehabilitation efforts were shifted to Midway Islands in 1992.

From May 1992 to January 1993, 20 juvenile females (1-, 2-, and 3-year-olds) were transported to Midway Islands. These seals, all in poor condition, were taken from French Frigate Shoals and either went directly to Midway or were taken there after a period of rehabilitation on Oahu. Of these 20, 2 died prior to release (one was unable to recover from its emaciated condition and one died from aspiration pneumonia). Four other seals died after release, but the cause(s) of death could not be determined due to the degree of decomposition.

The primary objective in transferring seals to Midway is to increase their chance for survival to reproductive maturity. Only seals in poor condition (and therefore not likely to survive) are moved. Their survival will be monitored closely to determine the success of rehabilitation and adaptation to their new environment. Secondarily, the release of these seals should enhance the recovery of the Midway Islands population. Currently, efforts are underway to identify the population of resident seals and thereby provide a baseline for measuring changes due to the release of rehabilitated seals at this site.

Necker Island. The Necker Island population of Hawaiian monk seals is small and may be limited by the lack of haulout area. Most of this island's shoreline is rocky and inaccessible. Where seals can come ashore, they are found in relatively high density. The nearshore environment is turbulent and does not provide the shallow protected waters characteristic of other monk seal haulout sites. These conditions appear to be less than suitable for the successful rearing of pups (Westlake and Gilmartin 1990), and probably limit the growth of this population.

In spite of these limits, counts at this site increased substantially after 1975 and remained at least stable, if not increasing slightly, through the 1980s (Fig. 27). Between 1983 and 1991, the mean of 19 counts was 28 (sd ± 6.7). If the fraction of seals hauled out at any given time during this period ranged from approximately 0.3 to 0.4 (Gilmartin et al., in press), then the Necker Island population could include on the order of 70 to 95 seals.

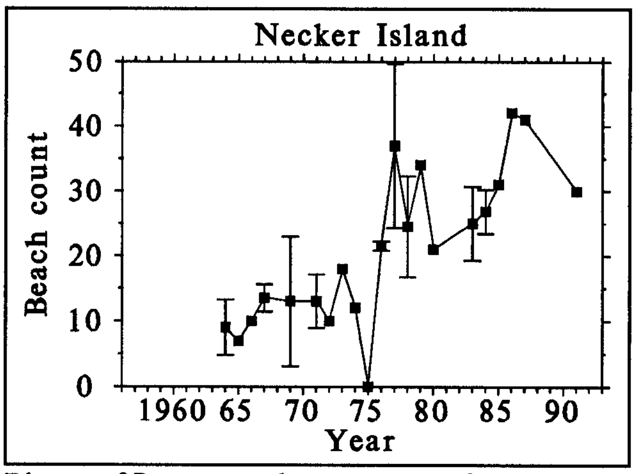

Figure 27. Beach counts of Hawaiian monk seals at Necker Island, 1956-92. See Appendix B for data and sources used to reconstruct the trend.

The growth of this population in the late 1970s was probably due to immigration rather than an increase in births. The first documented birth at this site was in 1977 (Giezentanner and Woodside 1977), and while monitoring has been sporadic and opportunistic, the maximum number of pups observed at Necker Island was 4 in 1982 (Conant 1985). Hence, a large portion of this population may be immigrants, particularly from French Frigate Shoals (approximately 150 km to the west; Fig. 1). In 1989, 40 seals were counted at Necker Island, of which 5 were originally tagged at French Frigate Shoals. Similarly, 30 seals were counted in 1991 and, again, 5 were known to be from French Frigate Shoals.

The increase of seals at Necker Island is consistent with the growth of the French Frigate Shoals population between the late 1950s and 1980s. The relationship of these two populations, and particularly the movement of seals from French Frigate Shoals to Necker Island, may be important to an understanding of the recent decline at French Frigate Shoals. Whenever possible, monitoring efforts will be increased to assess this migration and the status of the Necker Island population.

Nihoa Island. The Hawaiian monk seal population at Nihoa Island is small and, as at Necker Island, is expected to be limited by the lack of suitable habitat for hauling out and rearing pups. The shoreline is rocky and mostly inaccessible, and nearshore conditions are generally harsh.

Nevertheless, counts at this site increased substantially in the late 1970s and have remained relatively high since then (Fig. 28). The increase corresponds to the growth seen at Necker Island and, here again, is probably due to immigration. In 1991, 8 seals identified at this site were tagged at French Frigate

Shoals. At least 7 pups were born at Nihoa Island in 1991 and 5 in 1992, but sporadic observations in previous years suggest that pup production has been much lower (1-2 pups annually). Additional monitoring at Nihoa Island should provide information on the extent of movement from other populations (particularly French Frigate Shoals), and the dynamics of those populations.

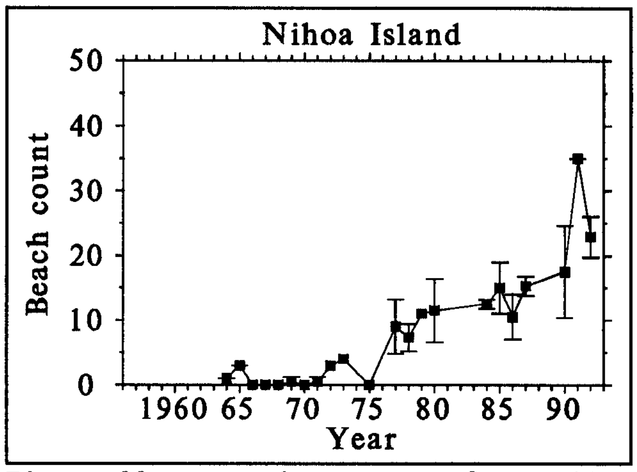

Figure 28. Beach counts of Hawaiian monk seals at Nihoa Island, 1956-92. See Appendix B for data and sources used to reconstruct the trend.

Growth Rate at MNPL

Hawaiian monk seal population growth has not been observed with sufficient frequency and reliability to estimate MNPL (maximum net productivity level) or the growth rate at MNPL. In addition, life history parameters and changes in those parameters with changes in density are not sufficiently known to model growth rate at MNPL. Hence, this growth rate is undetermined.

STOCK STATUS RELATIVE TO OSP AND K

Status under the Endangered Species Act and the Marine Mammal Protection Act

In 1976, the Hawaiian monk seal was designated as depleted under the Marine Mammal Protection Act of 1972 (41 FR 30120) and endangered under the U.S. Endangered Species Act of 1973 (41 FR 51611). The legal status of this species has not changed since those designations.

Methods of OSP Determination

OSP has been interpreted as a population level between K, the environmental carrying capacity, and MNPL, the maximum net productivity level (Federal Register, 21 December 1976, 41FR55536). Neither K nor MNPL has been estimated for the Hawaiian monk seal. Historical records indicate that marked changes have occurred in both total abundance and abundance at the individual islands/atolls, but the nature of those changes (i.e., endpoints and shapes of the growth curves and declines) cannot be characterized with sufficient reliability to estimate K or MNPL. Evidence suggests that the population at French Frigate

Shoals may be at or near K but, at present, this level cannot be estimated reliably for the remaining populations.

Similarly, MNPL cannot be estimated for this species. The primary methods of determining MNPL include empirical monitoring of growth from below to above MNPL, and modeling of such a growth curve based on knowledge of life table parameters and changes in those parameters with increasing density. A period of growth was observed for the population at French Frigate Shoals, but that growth was not monitored with sufficient frequency and reliability to determine MNPL. Also, the extent of immigration into the French Frigate Shoals population during that period has not been determined, and significant immigration would confound the estimation of MNPL. The current population at Pearl and Hermes Reef appears to be in the early phase of a growth curve, but is still well below its expected MNPL. Furthermore, available life history information is insufficient to evaluate the nature of density dependence in monk seal life table parameters and population growth cannot be modeled or predicted with sufficient confidence to estimate MNPL. Hence, both K and MNPL remain undetermined for this species.

In a management context, K and MNPL are intended to be measures of a population under <u>natural</u> conditions, where neither growth nor regulation is constrained by human-related factors. In the past, human activities have significantly influenced the demography of Hawaiian monk seal populations. Whether such influence continues at present is not clear, but the lack of recovery at some sites indicates that extraneous factors may be constraining growth. If such factors cannot be identified, then future efforts to estimate MNPL or K will be required to assume that those factors are no longer significantly affecting natural growth and regulation. Such an assumption would lessen confidence in the resulting estimates.

Condition Indices

Growth patterns of pups and juveniles are the only measures of condition under evaluation. Length and girth have been routinely measured for newly weaned pups in all monitored populations. At Laysan Island and French Frigate Shoals, pups and 1- and 2-year-old juveniles have been measured for length and axillary girth, and have also been weighed. These measures have been used to contrast condition of young seals at different sites and in different years. The additional measures at Laysan Island and French Frigate Shoals allow comparison of size and growth patterns for populations considered to be below K (Laysan Island) and near K (French Frigate Shoals). Preliminary results are consistent with the hypothesis that growth rates are lower at French Frigate Shoals and that these indices may therefore be useful for identifying populations at or near K. However, extensive interannual variation and small sample sizes argue that these growth patterns cannot be characterized with sufficient precision to be useful in determining MNPL.

HUMAN-RELATED MORTALITY AND REMOVALS

Permitted Take

Permitted takes of Hawaiian monk seals are allowed only for scientific and management purposes. Since 1982, 22 mortalities occurred during permitted activities, but the majority of those seals were undergoing rehabilitation because they were in poor condition and their chance of survival was significantly compromised. From 1983 to 1993, 51 weaned female pups were captured for rehabilitation and eventual release into the wild. Eleven (22%) died in captivity. In addition, since 1991, 2 subadult females, 11 juvenile females, and 2 juvenile males were taken into captivity; 3 juvenile females and the 2 juvenile males died before recovery and release. Other deaths involved 2 adult males and 1 juvenile male during separate research activities, an adult male being held for transport from Laysan Island to Johnston Atoll, an adult male transported to Honolulu for permanent captivity, and an adult male intentionally sacrificed because it was killing pups at French Frigate Shoals.

In addition to these mortalities, 8 adult males, 1 adult female and 1 subadult female are currently in captivity and will not be released into the wild population.

Incidental Take

Hawaiian monk seals are known to interact directly with lobster, bottomfish, and pelagic longline fisheries. Only a single incidental death has been documented; that death occurred in 1986 when a seal became entangled in the rope bridle of an actively fishing lobster trap. A number of additional, harmful interactions have been documented (Appendix C), but the rate and overall population effects of monk seal/fishery interactions are difficult to evaluate. Not all interactions are observed (e.g., some occur at night or with untended gear), and not all observed interactions are reported (for example, see Dollar 1992). A variety of interactions occur, ranging from the incidental death just mentioned to seals following fishing vessels and feeding on discarded catch or baitfish. Anecdotal reports suggest that monk seals are feeding on kahala (*Seriola dumerili*) discarded from bottomfishing operations. This fish is known to carry high levels of ciguatoxin and is therefore not sold for human consumption.

Subsistence Take

Legally, Hawaiian monk seals cannot be killed for subsistence. In a 1991 court case (United States of America versus Daniel Peter Kaneholani, United States Court of Appeals, 9th Circuit, No. 90-10643, filed September 18, 1991), a resident of Kauai was tried and convicted for killing an adult female monk seal. In spite of the resident's claim that as a native Hawaiian he had a right to kill the seal for subsistence, the court ruled that the killing was illegal.

Illegal Killing

The only known case of illegal killing is the incident just described, where a resident of Kauai killed an adult female seal. The observation of severe head injuries on several monk seals in 1991 (during the rapid expansion of pelagic longlining operations) suggests that seals may have died from such injuries, but evidence has not been found to substantiate this hypothesis.

Entanglement in Marine Debris

In 1992, 14 seals were found entangled in marine debris. Eight of these cases occurred at Lisianski Island, 2 each at French Frigate Shoals and Laysan Island, and 1 each at Kure Atoll and Pearl and Hermes Reef. The rate of entanglement (Fig. 26) has increased in the past 3 years, and is comparable to rates seen in 1987-89, when entanglement was at its highest recorded level.

Henderson (1990) reviewed incidents of entanglement through 1988. He found that pups were involved in approximately 42 percent of observed entanglements even though they comprise only about 11 percent of the total population. He also found that, as in 1992, the rate of entanglement (corrected for observation effort) was much greater at Lisianski Island than at other locations.

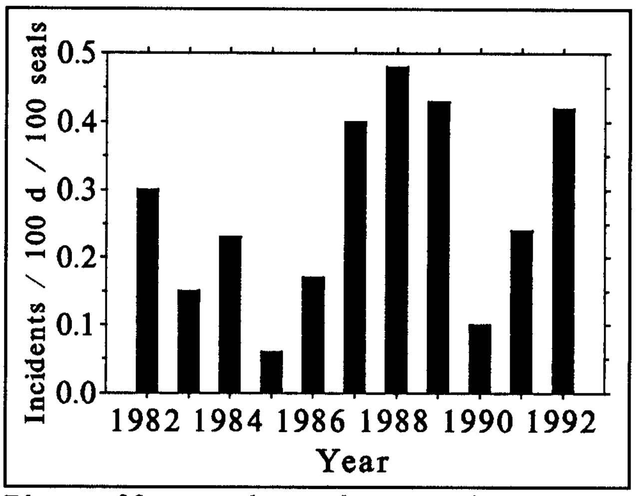

Figure 29. Number of entanglements of Hawaiian monk seals in marine debris, 1982-92. Data are corrected for number of seals seen and length of observation period.

The total loss of seals due to entanglement is difficult to measure because an undetermined number of entanglements occur at sea where they are not observed. The coral reef habitat of Hawaiian monk seals further increases the encounter rate of seals with net debris because such debris is frequently caught on the reef and may continue to pose a threat to seals for extended periods of time. Whenever possible, potentially entangling debris are collected, sampled, weighed, and destroyed at each field site. Total weights of this debris for 1987-91 are given in Table 4.

Unknown and Other Causes

A number of potential human-related sources of Hawaiian monk seal mortality are difficult to assess or quantify. For example, in the 1960s, the effects of disturbance at Kure Atoll were relatively clear (Kenyon 1972). At present, some populations are subjected to lower levels of disturbance and the effects, if any, are not clear.

Direct interactions of seals with fisheries are of obvious concern and, in comparison with indirect interactions, should be relatively easy to measure. Indirect interactions, particularly competition for prey, are not measurable at the present time, but may be very important when populations are food-limited, as appears to be the case at French Frigate Shoals.

Additional human-related sources of Hawaiian monk seal mortality include construction and other debris left at certain locations in the NWHI. For example, in 1986 at East Island, French Frigate Shoals, a weaned pup drowned after becoming entangled in wiring left by the Coast Guard, who had abandoned the island approximately 3 decades earlier (Henderson 1990). And in 1991 a seal died after becoming entrapped behind the decaying seawall at Tern Island, French Frigate Shoals. Extensive military debris at Eastern Island (Midway Islands) also poses a threat to seals and other wildlife. Other hazards for monk seals may include various pollutants or toxins left on certain islands

Table 4. Total weight (kg) of potentially entangling marine debris by island/atoll for 1987-91 (NMFS unpubl. data).

Year	FFS	Laysan	Lis	P&H	Kure	Total
1987	447	601	---	939	3,600	5,587
1988	71	553	---	428	1,206	2,258
1989	871	742	---	417	2,383	4,414
1990	393	883	4,074	595	1,329	7,274
1991	497	1,603	3,271	1,515	1,424	8,310
Total	2,281	4,382	7,345	3,895	9,940	27,843

(i.e., Tern Island at French Frigate Shoals, Midway Islands) and leaching into the nearshore environment. The extent to which such contaminants affect Hawaiian monk seals is undetermined.

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APPENDIX A - FISHERY MANAGEMENT PLANS

A brief outline of fishery management plans as they relate to the management of Hawaiian monk seals.

I. Combined Fishery Management Plan, Environmental Impact Statement, Regulatory Analysis and Draft Regulations for the Spiny Lobster Fisheries of the Western Pacific Region (May 1981).

A. Actions:

- Establishes a minimum size limit for harvested spiny lobsters.
- Requires that only traps be used to harvest lobsters.
- Requires that all egg-bearing and small (below legal size) lobsters be released with a minimum of injury.
- Prohibits spiny lobster fishing in the FCZ of the NWHI in waters less than 10 fathoms deep and in the FCZ less than 20 miles from Laysan Island.
- Requires commercial fishermen to obtain permits and submit reports on catch and effort data.
- Provides a mechanism for verification, investigation, evaluation, and response to any report of an incident involving the mortality of a monk seal.

B. Amendments:

- #1 (June 1983). Adopts the State measures for the lobster fishery around the main Hawaiian Islands.
- #2 (August 1983). Modifies the existing regulations regarding size specifications for lobster trap openings.
- #3 (October 1985). Redefines the measurement criterion used to distinguish legal from undersized lobsters, and eliminates the allowance for a percentage of undersized lobsters in the catch.
- #4 (October 1986). Prohibits fishing for all lobsters (spiny and slipper) in areas where

previous regulations only prohibited fishing for spiny lobsters.

- #5 (September 1987). Establishes a minimum legal size for the slipper lobster, requires escape vent panels for all lobster traps, requires release of all egg-bearing lobsters regardless of species, revises the daily lobster catch report and permit application forms, eliminates the annual processor report, revises the trip processing and sales report, and changes the name from Spiny Lobster Fishery Management Plan to Crustacean Fishery Management Plan.

- #6 (October 1990). Defines recruitment overfishing to occur "when the spawning potential ratio (measured for a specific fishing area) is 0.2 or below."

- #7 (October 1991). For the NWHI fishery, establishes a limited access system with limited access permits and gear restrictions. Also establishes an adjustable annual quota and annual closed season.

II. Combined Fishery Management Plan, Environmental Assessment and Regulatory Impact Review for the Bottomfish and Seamount Groundfish Fisheries of the Western Pacific Region (March 1986).

A. Actions:

- Prohibits the use of bottom trawls and bottom-set gillnets for commercial harvest of bottomfish in the FCZ.

- Establishes a 6-year moratorium on fishing in the FCZ at the Hancock Seamounts.

- Prohibits the use of explosives and poisons for harvesting groundfish in the FCZ

- Requires a permit for bottomfishing in the FCZ of the NWHI.

- Provides for an experimental fishery permit only available to domestic operators.

- Requires annual review of the fishery, and utilizes existing State and Territory reporting systems for data collection.

B. Amendments:

- #1 (June 1987). Provides for limited access to fishing for bottomfish in the U.S. EEZ around American Samoa and Guam, and extends the due date for the Annual Report from 31 March to 30 June.

- #2 (March 1988). Provides for limited access to the bottomfish fishery of the NWHI.

- #3 (October 1990). Defines recruitment overfishing for bottomfish species as occurring ". . . when the Spawning Potential Ratio (SPR; Goodyear 1989), (i.e., the ratio of the spawning stock biomass per recruit at the current level of fishing ($SSBR_f$) to the spawning stock biomass per recruit that would occur in the absence of fishing ($SSBR_u$)), is equal to or less than .20."

- #4 (May 1991). Establishes protected species zones within 50 nmi of French Frigate Shoals, Gardner Pinnacles, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Islands, Kure Atoll, Nihoa Island, Necker Island, and Maro Reef. Operators intending to fish within these zones must notify NMFS prior to such fishing, and must accommodate and observer if so requested. The regional director can alter the sizes of the protected species after consultation with the Western Pacific Regional Fishery Management Council. This amendment also modifies the permit application form and prevents overlapping permits for the Mau and Ho'omalu Zones in the NWHI.

III. Fishery Management Plan for the Pelagic Fisheries of the Western Pacific Region (July 1986).

A. Actions:

- Prohibits foreign longline vessels from fishing within 100 nmi of the NWHI (west of 161°W longitude) including Midway Island.

- Requires foreign longline vessels to file effort plans and obtain permits prior to fishing in the remaining open areas of the FCZ, and then to collect and submit to NMFS data on catch and effort and on sea turtle and marine mammal interactions.

- Requires foreign longline vessels to carry observers when directed by NMFS.

- Prohibits drift gillnetting by foreign or domestic operators anywhere in the FCZ of the Western Pacific Region.

- Allows domestic vessels to fish with drift gillnets in the FCZ with an experimental fishery permit, but requires such vessels to collect and submit data on catch and effort and interactions with sea turtles and marine mammals.

B. Amendments:

- #1 (November 1990). Includes a definition of overfishing and revises the definition of Optimum Yield.

- #2 (February 1991). (1) Defines the management units for these fisheries on the basis of individual species fished, and defines the management area to include fishing and support activities that occur outside the FCZ but affect the fishery and its management within the FCZ. Also requires a permit for any U.S. longline vessel fishing or transhipping longline-caught fishes within the EEZ of the Western Pacific Region, or landing longline-caught fishes in the U.S. and its possessions in the Pacific. (2) Requires domestic longliners to keep daily records of fishing effort and catches and observations or encounters with protected species. (3) Requires domestic longline vessels to notify NMFS prior to fishing within a 50-mile protected species study area around Nihoa Island, Necker Island, French Frigate Shoals, Gardner Pinnacles, Maro Reef, Laysan Island, Lisianski Island, Pearl and Hermes Reef, Midway Island, and Kure Atoll of the NWHI. Vessels in the study area must carry observers if required. (4) Requires operators of longline vessels to attend an orientation meeting to learn about procedures for protecting endangered and threatened species.

- #3 (June 1991). Designates a Protected Species Zone consisting of all waters within 50 nmi of the islands and atolls of the NWHI from Kure Atoll to Nihoa Island, including the corridors between these islands. Prohibits longline fishing within this zone. Establishes a process by which the regional director of NMFS can designate conservation and management measures necessary to safeguard protected species.

- #4 (June 1991). Extends the Hawaii longliner moratorium for a total 3 years. (An emergency moratorium was established on 23 April, 1991.)

- #5 (October 1991). Incorporates into the fishery management plan a moratorium on longline fishing around the main Hawaiian Islands, and provides the mechanism for changing the size of affected areas.

APPENDIX B - POPULATION TREND DATA AND SOURCES

Tables of data sources used to approximate the 1956-92 trends of Hawaiian monk seal populations. Full citations for sources are listed at the end of the tables. Unless noted, only complete ground counts between 1 March and 30 September were used.

Table B1. Data and sources used to reconstruct the 1956-92 population trend at French Frigate Shoals (Fig. 9 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1957	1	35[a]	-	Fiscus et al. 1978
1958	1	43[b]	-	Rice 1960
1962	1	27[a]	-	Kramer and Beardsley 1962
1963	1	55[c]	-	Amerson and Sibley 1963
1969	1	159	-	Kridler 1969a
1970	1	166[a]	-	Kridler 1970b
1971	2	155.5[a]	9.2	Olsen 1971a,b
1972	1	204[d]	-	Olsen 1972
1973	1	206[d]	-	Olsen 1973b
1975	1	274[e]	-	FWS unpubl. data
1976	1	195	-	DeLong 1976, DeLong et al. 1976
1977	5	185.4	-	DeLong and Brownell 1977, Kenyon and Rauzon 1977, Rauzon et al. 1978
1978	2	198.0	1.4	Fiscus et al. 1978
1979	1	241	-	Rauzon 1979
1980	16	241.1[f]	20.5	Johnson and Johnson 1984
1981	3	216.3[a]	4.0	Rauzon 1981
1983	1	218	-	NMFS unpubl. data
1984	4	221.0[g]	28.1	Eliason and Henderson 1992
1985	9	313.4	44.1	NMFS unpubl. data
1986	3	321.3	69.1	NMFS unpubl. data
1987	9	318.9	35.4	NMFS unpubl. data
1988	8	311.4	23.5	Craig et al. 1993
1989	15	335.9	29.8	Craig et al. 1993

Table B1. Continued.

Year	Number counts	Mean count	St.dev.	Source(s)
1990	9	294.6	23.2	NMFS unpubl. data
1991	10	221.9	40.5	NMFS unpubl. data
1992	11	224.1	33.1	NMFS unpubl. data

[a] Count from aerial survey.

[b] Some counts from boat; not clear that all islets counted and therefore minimum count.

[c] Count not exact; more than 55 seals (minimum count).

[d] Count from combination ground/aerial survey.

[e] Type and completeness of count unknown.

[f] Estimate based on the sum of 3 means: 1) 195.9 (sd ± 16.5, 16 counts) for all islands except Disappearing and Shark Islands, 2) 32.1 (sd ± 8.4, 9 counts) for Disappearing Island, and 3) 13.1 (sd ± 8.9, 11 counts) for Shark Island.

[g] Counts from January, no atoll counts made in March-September 1984.

Table B2. Data and sources used to reconstruct the 1956-92 population trend at Laysan Island (Fig. 12 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1958	2	317	12.7	Rice 1960, 1990
1959	3	223.3	11.0	Ely and Clapp 1973, Smythe 1960, Udvardy 1961
1960	1	108	-	Ely and Clapp 1973
1961	2	224.5	6.4	Woodside 1961, Woodside and Kramer 1961
1962	1	261	-	Kramer and Beardsley 1962, Marshall 1962
1964	2	282	42.4	Amerson 1964, Kridler 1964a, Walker 1964
1965	2	227	24.0	Wirtz 1965, Ely and Clapp 1973
1966	4	212.3	21.3	Kenyon 1966; Kridler 1966a,b; Ely and Clapp 1973
1967	3	156	59.0	Kridler 1967a,c; Ely and Clapp 1973
1968	2	180.5	2.1	Kridler 1968a,b; Ely and Clapp 1973
1969	3	180.3	32.1	Kridler 1969a,b; Olsen 1969
1970	2	140.5	9.2	Kridler 1970a
1971	1	239	-	Kridler 1971a
1972	1	197	-	Kridler 1972a
1973	1	157	-	Olsen 1973a
1974	1	186	-	Sincock 1974, Sincock et al. 1974
1975	1	139	-	Sekora 1975, Sincock 1975
1976	5	168.6	23.1	DeLong 1976, DeLong et al. 1976, Sekora and Geizentanner 1976, Yuen 1976
1977	52	177.9	18.8	DeLong and Brownell 1977; Fiscus 1977; Giezentanner 1977, 1978; Johnson and Johnson 1978
1978	79	124.6	12.9	Johnson and Johnson 1981a
1979	81	112.6	14.8	Johnson and Johnson 1980, Rauzon 1979
1980	89	117.6	15.6	Harrison 1980, Johnson and Johnson 1981b

Table B2. Continued.

Year	Number counts	Mean count	St.dev.	Source(s)
1981	12	113.3	15.1	Knudtson 1981, Rauzon 1981
1982	48	106.3	11.3	Alcorn 1984
1983	24	95.2	11.7	Alcorn and Buelna 1989
1984	45	98.7	9.7	Johanos et al. 1987
1985	69	116.4	11.9	Becker et al. 1989
1986	20	144.4	12.8	NMFS unpubl. data
1987	26	147.1	19.6	NMFS unpubl. data
1988	26	110.4	13.5	Johanos, et al. 1990
1989	26	100.5	12.4	NMFS unpubl. data
1990	39	80.9	12.8	NMFS unpubl. data
1991	37	88.1	9.3	NMFS unpubl. data
1992	27	92.3	13.0	NMFS unpubl. data

Table B3. Data and sources used to reconstruct the 1956-92 population trend at Lisianski Island (Fig. 17 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1957	1	256[a]	-	Olsen 1971b
1958	1	256[a]	-	Rice 1960
1961	1	172	-	Woodside and Kramer 1961
1963	1	210	-	Wirtz 1966, Clapp and Wirtz 1975
1964	3	148.3	29.7	Amerson 1964, Kridler 1964a, Wirtz 1966, Clapp and Wirtz 1975
1965	2	174	18.4	Wirtz 1965, 1966; Clapp and Wirtz 1975
1966	2	148	12.7	Kridler 1966a, Clapp and Wirtz 1975
1967	4	129	15.1	Kridler 1967a, Clapp and Wirtz 1975
1968	1	123	-	Kosaka 1968a, Kridler 1968b
1969	2	128.5	2.1	Kridler 1969b, Olsen 1969, Clapp and Wirtz 1975
1970	1	109	-	Kridler 1970a
1971	1	119	-	Kridler 1971a
1972	1	80	-	Kridler 1972a
1973	3	96	6.2	Kridler 1973a, Olsen 1973a
1974	1	96	-	Sincock 1974
1975	1	95	-	Sekora 1975, Sincock 1975
1976	3	99.7	30.4	DeLong 1976, DeLong et al. 1976
1977	6	84.8	16.6	DeLong and Brownell 1977, Giezentanner and Kridler 1977, Giezentanner 1978
1978	10	76.5	8.8	Fiscus et al. 1978, Harrison 1978
1979	4	80	15.6	Rauzon 1979
1980	23	87.2	28.7	NMFS unpubl. data
1981	1	88	-	NMFS unpubl. data
1982	85	97.3	10.2	Stone 1984
1983	52	100.6	11.4	Johanos and Kam 1986
1984	16	75.4	8.8	Alcorn et al. 1988
1985	15	88.9	11.3	Alcorn et al. 1988

Table B3. Continued.

Year	Number counts	Mean count	St.dev.	Source(s)
1986	10	104.9	11.1	Westlake and Siepmann 1988
1987	11	98.6	9.4	Johanos and Withrow 1988
1990	22	81	10.0	NMFS unpubl. data
1991	12	81.7	7.4	NMFS unpubl. data
1992	28	82.7	11.6	NMFS unpubl. data

[a] Count from aerial survey.

Table B4. Data and sources used to reconstruct the 1956-92 population trend at Pearl and Hermes Reef (Fig. 20 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1957	2	171.5[a]	14.8	Rice 1990
1958	3	200.7[a]	13.5	Rice 1990
1963	1	201	-	Wirtz 1966
1964	2	123.0	2.8	Amerson 1964, Wirtz 1966, Amerson et al. 1974
1965	1	175	-	Wirtz 1966, Amerson et al. 1974
1970	1	122	-	Olsen 1970
1971	1	106	-	Kridler 1971b
1972	1	61	-	Kridler 1972b
1973	2	35.0[b]	1.4	Kridler 1973a,b
1974	1	35[b]	-	Kridler 1974
1976	1	26	-	DeLong 1976, DeLong et al. 1976
1977	1	42	-	DeLong and Brownell 1977, Giezentanner 1978
1978	2	24.5	2.1	Fiscus et al. 1978, Harrison 1978
1979	1	28	-	Rauzon 1979
1981	1	41[a]	-	Rauzon 1981
1982	[8,14]	55.7[c]	8.7	Forsyth et al. 1988
1983	5	31.4	8.2	NMFS unpubl. data
1984	5	39.6	10.5	NMFS unpubl. data
1985	12	53.9	8.6	NMFS unpubl. data
1986	6	54.2	6.1	NMFS unpubl. data
1987	11	43.0	10.6	NMFS unpubl. data
1988	14	62.6	9.9	Choy and Hiruki 1992
1990	2	47.5	10.6	Finn et al. in prep
1991	11	53.6	9.8	Finn et al. in prep

Table B4. Continued.

Year	Number counts	Mean count	St.dev.	Source(s)
1992	5	88.8	13.5	NMFS unpubl. data

[a] Count from aerial survey.

[b] Counts from helicopter surveys.

[c] Based on the sum of mean counts for individual islands. From August 10-25, 8-14 counts at each island.

Table B5. Data and sources used to reconstruct the 1956-92 population trend at Kure Atoll (Fig. 23 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1957	1	128[a]	-	Kenyon and Rice 1959
1958	3	94.0	41.6	Rice 1960, Kenyon 1966
1959	1	53	-	Woodward 1972
1960	1	60	-	Woodward 1972
1961	1	48	-	Walker 1979
1964	1	69	-	Walker 1964, 1979
1965	11	50.5[b]	24.2	Woodward 1972
1966	9	26.6[b]	12.3	Woodward 1972
1967	1	59	-	Walker 1979
1968	5	51.4	13.8	Kosaka 1968a, Kenyon 1976
1969	1	41	-	Walker 1969a,b
1972	1	33	-	Kosaka 1972, Kridler 1972a
1973	1	18	-	Telfer 1973
1974	1	25	-	Walker 1979
1975	1	47	-	Iverson 1975
1976	3	35.0[c]	6.0	DeLong et al. 1976, Woodside and Okamoto 1976, Walker 1979
1977	1	27	-	DeLong and Brownell 1977
1978	3	25.0	7.0	Harrison 1978, Walker 1979
1979	2	34.5	9.2	Rauzon 1979
1981	21	26.0	5.8	Gilmartin et al. 1986
1982	42	27.8	5.9	Bowlby et al. 1991
1983	32	24.1	5.7	Bowlby et al. 1991
1984	52	25.1	5.0	NMFS unpubl. data
1985	67	26.1	5.6	Reddy and Griffith 1988
1986	15	28.4	3.8	NMFS unpubl. data
1987	62	27.0	7.3	Reddy 1989
1988	54	30.0	6.7	Henderson and Finnegan 1990
1989	54	29.2	7.5	NMFS unpubl. data

Table B5. Continued.

Year	Number counts	Mean count	St.dev.	Source(s)
1990	44	29.5	5.9	NMFS unpubl. data
1991	44	33.8	10.3	NMFS unpubl. data
1992	15	41.3	7.2	NMFS unpubl. data

[a] Count from aerial survey.

[b] Sand Island counted from Green Island.

[c] Count based on combined aerial and ground survey.

Table B6. Data and sources used to reconstruct the 1956-92 population trend at Midway Islands (Fig. 26 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1957	3	53.3	9.6	Kenyon 1972
1958	1	52	-	Kenyon 1972
1968	1	4	-	Kenyon 1972
1972	1	0	-	Kridler 1972a
1975	1	5	-	Iverson 1975
1977	2	4.5	0.7	DeLong and Brownell 1977, Giezentanner 1978
1978	1	3	-	Fiscus et al. 1978
1979	1	3	-	Rauzon 1979
1980	1	0	-	Harrison 1980
1983	1	8	-	NMFS unpubl. data
1985	1	3	-	NMFS unpubl. data
1986	1	5	-	NMFS unpubl. data
1987	1	2	-	NMFS unpubl. data
1989	4	4.8	-	NMFS unpubl. data
1990	11	5.3	2.0	NMFS unpubl. data
1991	7	5.6	2.1	NMFS unpubl. data
1992	8	9.4	3.5	NMFS unpubl. data

Table B7. Data and sources used to reconstruct the 1956-92 population trend at Necker Island (Fig. 27 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1964	2	9	4.2	Kridler 1964a,b
1965	1	7	-	Kridler 1965, Banko 1965
1966	1	10	-	Kenyon 1966
1967	2	13.5	2.1	Clapp and Kridler 1977, Kridler 1967a,b
1969	2	13	9.9	Kridler 1969b, Olsen 1969
1971	2	13	4.1	Kridler 1971a, Olsen 1971a
1972	1	10	-	Kridler 1972a
1973	1	18	-	Olsen 1973a
1974	1	12	-	Vailati 1974
1975	1	0	-	Sekora 1975
1976	2	21.5	0.7	Sekora and Giezentanner 1976, Yuen 1976
1977	2	37	12.7	Giezentanner and Woodside 1977, Giezentanner 1978
1978	2	2.5	7.8	Fiscus et al. 1978, Harrison 1978
1979	1	34	-	Rauzon 1979
1980	1	21	-	Harrison 1980
1983	9	25	5.7	Conant 1985, Morrow and Buelna 1985
1984	6	26.8	3.4	NMFS unpubl. data
1985	1	31	-	Becker 1985
1986	1	42	-	NMFS unpubl. data
1987	1	40	-	NMFS unpubl. data
1991	1	30	-	NMFS unpubl. data

Table B8. Data and sources used to reconstruct the 1956-92 population trend at Nihoa Island (Fig. 28 in text).

Year	Number counts	Mean count	St.dev.	Source(s)
1964	2	1	0.0	Amerson 1964, Kridler 1964a
1965	2	3	0.0	Banko 1965, Kridler 1965
1966	1	0	-	Clapp et al. 1977
1967	2	0	0.0	Clapp et al. 1977
1968	2	0	0.0	Clapp et al. 1977
1969	2	0.5	0.7	Clapp et al. 1977
1970	1	0	-	Kridler 1970a
1971	2	0.5	0.7	Kridler 1971a, Clapp et al. 1977
1972	1	3	-	Kridler 1972a
1973	1	4	-	Olsen 1973a
1975	1	0	-	Sekora 1975
1977	2	9.0	4.2	Giezentanner and Woodside 1977
1978	3	7.3	2.1	Harrison 1978
1979	1	11	-	Rauzon 1979
1980	1	11.5	4.9	Harrison 1980, Conant 1983
1984	2	12.5	0.7	NMFS unpubl. data
1985	3	15.0	4.0	NMFS unpubl. data
1986	2	10.5	3.5	NMFS unpubl. data
1987	3	15.3[a]	1.5	NMFS unpubl. data
1990	2	17.5	7.1	NMFS unpubl. data
1991	2	35.0	0.0	NMFS unpubl. data
1992	8	22.9	3.1	NMFS unpubl. data

[a] Counts incomplete; west ledge (sector 5) not included.

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APPENDIX C - HAWAIIAN MONK SEAL/FISHERY INTERACTIONS

Known or probable interactions of monk seals with fisheries in the Northwestern Hawaiian Islands. Table limited to incidents in which a seal was injured or, in one case, killed.

Date	Location	Seal	Observations
-/82	French Frigate Shoals	subadult, female	Seal with bottomfish hook in mouth; seen subsequently without hook
9/85	Kure Atoll	pup, female	Seal with fish hook initially in mouth, then chest; hook removed[a]
11/86	Necker Island		Seal found dead, entangled in bridle rope over actively fishing lobster trap
-/89	Kauai	yearling, female	Seal with large hook in mouth; hook removed
-/89	French Frigate Shoals	4 adults, 3 subadults, 3 juveniles	Seals with probable propeller wounds[b]
5/90	French Frigate Shoals	4 adults, 2 subadults, 1 yearling	Seals with unusual head wounds
1/91	French Frigate Shoals	4 adults, 1 subadult, 1 juvenile, 3 unknown size	3 seals with hook in mouth or line protruding from mouth, 5 unusual head injuries, 1 unusual lateral injury
3/91	Kure Atoll	juvenile, female	Seal with fish hook in lower lip.
6/91	Kure Atoll	pup, female	Seal with fish hook in mouth; hook removed

[a] For description, see Reddy and Griffith (1988).
[b] Wounds are described in Craig et al. (1993).